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CIBA-GEIGY CRANSTON FACILITY
PHASE II REMEDIAL INVESTIGATION

ECOLOGICAL SCREENING ASSESSMENT
(Pawtuxet River)

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Prepared for:

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CIBA-GEIGY CORPORATION
444 SAW MILL RIVER ROAD
ARDSLEY, NY 10502-2699

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Prepared by:

IT Corporation
Knoxville, Tennessee
Edison, New Jersey
IT Project No. 408678.23.05

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EXECUTIVE SUMMARY

This screening-level ecological risk assessment was prepared by IT Corporation (IT) as part of a Resource Conservation and Recovery Act (RCRA) Facility Investigation (RFI) for the non-operational, and partially dismantled, CIBA-GEIGY facility ("Facility") at Cranston, Rhode Island. The main objective of this assessment was to evaluate risks posed to the Pawtuxet River ecosystem by particular constituents of concern, using existing data and conservative assumptions in order to focus future data collection efforts. Secondary objectives were determination of the advisability of fish tissue analyses, identification of probable sources of constituents of concern (COCs), and selection of analytes for the Phase II investigation.

The Facility is located in the town of Cranston, Rhode Island, approximately ten miles south of Providence, Rhode Island. The Pawtuxet River, which flows past the Facility on its way to Pawtuxet Cove, meanders through wooded areas, wetlands, commercial, industrial, and residential areas. This assessment covers the length of the river between Elmwood Avenue and Rhodes-on-the-Pawtuxet, including the Facility reach. The Pawtuxet has received discharges from many industries in the past and present. Before the industrial revolution (1800s) and dating back to the 1700s, forges and textile mills discharged to the Pawtuxet River; privies serving up to 3000 employees were positioned directly over the river. Currently, the waste water treatment plants of Warwick, West Warwick, and Cranston municipalities, as well as industrial operations, are upstream of the facility.

Problem Formulation involved determining surface water and sediment COCs, primary and secondary exposure pathways, potential ecological receptors, the potential for adverse effects due to the presence of COCs, and appropriate assessment and measurement endpoints. The COC selection process identified eight analytes in surface water COCs, versus 54 sediment COCs. Because bioassay testing was unable to identify significant mortality in test species exposed to surface water samples, this screening-level assessment focused only on COCs in sediments.

Potential primary exposure pathways for aquatic receptors in the river, and terrestrial receptors that utilize the river, include: (a) direct contact with contaminated sediments or surface waters, (b) uptake through roots in contact with surface water or sediments, (c) ingestion of contaminated surface waters, (d) incidental ingestion of contaminated sediments by either aquatic or terrestrial consumers, and (e) secondary exposure pathways for both aquatic and terrestrial receptors that involve ingestion of contaminants which have bioaccumulated into forage or prey items.

Terrestrial/riparian reconnaissance, fish population, and benthic invertebrate surveys were conducted at and near the Facility. Upland areas and riparian zones were found bordering the Pawtuxet River; a wetland area was observed downstream. White suckers (*Catastomus commersoni*) were numerically dominant at all areas surveyed. Common carp (*Cyprinus carpio*) were abundant, particularly near the Production Area end of the facility. Golden shiner (*Notemigonus crysoleucas*) were common. All other species collected were relatively few in number. The terrestrial survey identified twenty-eight species of upland plants and twenty-six species of riparian/wetland plants at and near the Facility. Twenty-six species

of birds were identified as well. These included the great blue heron (*Ardea herodias*), mallard duck (*Anas platyrhyncus*), and red-tailed hawk (*Buteo jamaicensis*). Five mammal species were identified, including the Eastern gray squirrel (*Sciurus carolinensis*) and the raccoon (*Procyon lotor*).

Problem Formulation resulted in a conceptual model whereby the Pawtuxet River at and below the Facility contains COCs in abiotic media at detectable concentrations and provides exposure pathways linking these COCs to both onsite and offsite ecological receptors. Fish and invertebrate species are directly exposed to COCs in surface water and sediments, while higher trophic level receptors may be exposed to COCs bioconcentrated in their prey species.

Exposure Characterization used simple equilibrium models to estimate potential exposures of generic invertebrates, generic fish, and a representative piscivorous species (great blue heron (*Ardea herodias*) to COCs in sediment pore waters. Effect Characterization was addressed using four approaches: (a) analysis of benthic invertebrate community structure, (b) results of bioassay testing of sediment, surface water, and pore water, (c) comparison, with a quotient methodology, of modeled exposure point concentrations to previously published effect levels for terrestrial and aquatic animals, and (d) other observed effects. Risk Characterization used a weight-of-evidence approach, wherein several qualitative and quantitative lines of evidence were integrated to summarize and interpret the ecological significance of any observed or predicted effects and the degree of risk posed to ecological receptors.

Results based on field surveys, bioassay tests, and simple ecotoxicological models suggest that conditions along the length of the Pawtuxet River investigated do not meet the assessment endpoint for benthic organisms and fish. COCs are present in sediments at concentrations potentially capable of inducing adverse affects; i.e., their toxicity quotient values are greater than zero. Some degree of chronic stress, most probably from chemical stressors, is evident in benthic invertebrate and fish populations throughout the length investigated. A few widely distributed, highly toxic, and non-Facility specific COCs (most notably copper) are undoubtedly responsible for at least some of the ecological stress observed in benthos and fish throughout the length investigated.

Ecotoxicological results produced by this screening assessment suggest that the Pawtuxet River has a high probability of meeting the assessment endpoints for wildlife species because the potential for adverse impacts in terrestrial, piscivorous species from the consumption of COCs bioaccumulated in fish prey was estimated to be minimal. These results strongly suggest that chemical analysis of fish tissue is not warranted. This position may be substantiated with more detailed food web modeling in the baseline ecological risk assessment.

Ecological values in the Pawtuxet River worthy of preservation or restoration could include a healthy, functioning benthic infauna and fish populations with normal demographic characteristics. Remedial actions taken to address sediments containing site-related contaminants in the Facility reach would contribute to the restoration of better ecological values in the Pawtuxet. However, the river ecosystem is unlikely to receive maximum benefits from any actions unless contaminant sources not related to the Facility are also addressed. These include: (a) contaminated sediments upstream of the Facility reach, (b)

waste water treatment plant and industrial discharges upstream of the Facility reach, and (c) non-point source discharges, such as storm runoff and atmospheric deposition, that enter the river at numerous points along the length investigated.

Based on the results of this screening assessment, it is recommended that analytes for Phase II investigations be limited to a group of eight "indicator" COCs. This will permit a more thorough examination the baseline ecological risk assessment for the Pawtuxet River of COCs that either make the greatest contribution to the overall potential for toxic effects in the river or are more clearly Facility-related or both.

1.0 INTRODUCTION

This screening-level ecological risk assessment was prepared by IT Corporation (IT) as part of a Resource Conservation and Recovery Act (RCRA) Facility Investigation (RFI) for the non-operational, and partially dismantled, CIBA-GEIGY facility at Cranston, Rhode Island. This report is based on the risk assessment process as defined by the "Framework for Ecological Risk Assessment" (USEPA, 1992a).

1.1 Background

The Cranston facility ("Facility") is located in the town of Cranston, Rhode Island, approximately ten miles south of Providence, Rhode Island. The Pawtuxet River flows past the facility on its way to Pawtuxet Cove (Figure 1-1). The Pawtuxet River has received discharges in the past, and continuing into the present, from a variety of industrial and municipal treatment plant sources. Currently, waste treatment plants for the municipalities of Warwick, West Warwick, and Cranston, as well as industrial metal plating operations, chemical manufacturers, and jewelry manufacturers, are upstream of Facility.

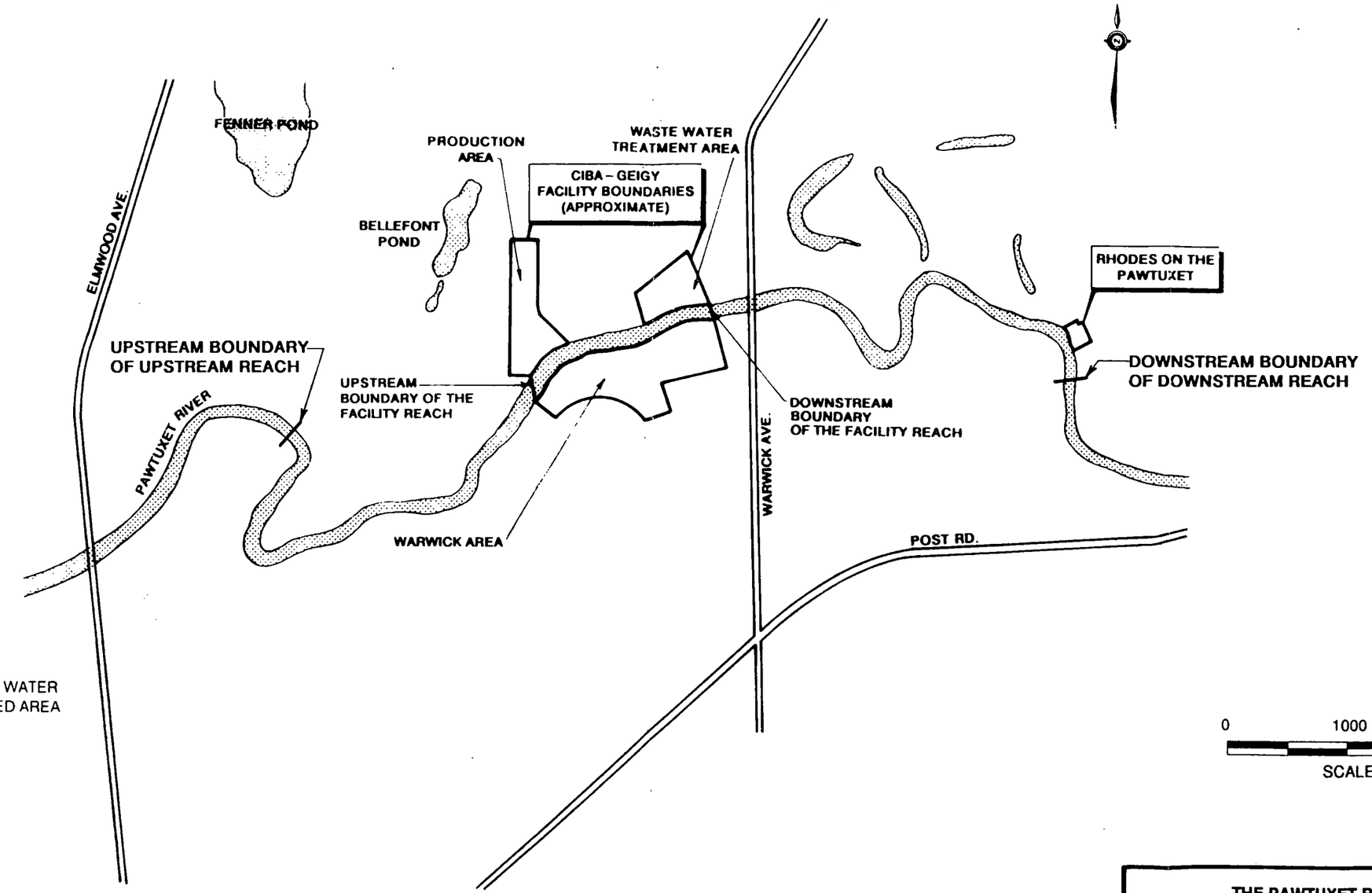
1.2 Objectives and Scope

The main objective of this assessment is to evaluate the potential risk of particular constituents of concern (COC) upon a section of the Pawtuxet River ecosystem. Specific objectives for this screening-level ecological risk assessment were to: (a) review existing and recently collected ecological data, (b) summarize this existing data into a description of ecological conditions at the Facility, (c) select constituents of concern (COC) based on physiochemical and ecologically relevant criteria, (d) develop a conceptual model to identify reasonable exposure pathways and potential ecological receptors, (e) make an initial assessment of the potential for COCs to induce adverse ecological effects, and (f) where adverse impacts are suggested but not quantifiable with available data, identify critical data gaps, define additional data requirements, and make recommendations for additional investigations (if any) required to support a baseline ecological risk assessment.

1.3 Organization

This report is organized in the following manner, which is consistent with the organization suggested by EPA guidance (USEPA, 1989b, 1991, 1992a):

Section 1.0	Introduction; outlines objectives and scope for this assessment and provides general information on the facility being investigated.
Section 2.0	Area Description; a brief discussion of ecological and ecologically-related resources and features in areas adjacent to the Pawtuxet River.
Section 3.0	Problem Formulation; involves developing a qualitative description of the potential for adverse effects and a clear definition of the problem(s) to be addressed by the assessment.
Section 4.0	Exposure Characterization; characterizes contaminant transport and fate phenomena, identifies site-specific ecological receptors, and quantifies exposure point concentrations from both primary and secondary exposure pathways.
Section 5.0	Ecological Effects Characterization, discusses quantitative links between contaminant concentrations and effects in receptors. Literature reviews are the primary source of such



LEGEND

MAJOR SURFACE WATER BODIES IN MAPPED AREA

0 1000 2000 FT

SCALE

BASE MAP SOURCE:

AERIAL PHOTOGRAPHS BY GEOD CORPORATION OF NEWFOUNDLAND, NEW JERSEY.
DATE FLOWN: 2 APRIL 1989.

DRAFT

THE PAWTUXET RIVER REACHES: UPSTREAM, FACILITY, AND DOWNSTREAM

WOODWARD – CLYDE CONSULTANTS

CONSULTING ENGINEERS, GEOLOGISTS AND ENVIRONMENTAL SCIENTISTS
WAYNE, NEW JERSEY

DR. BY:	KJFH	SCALE:	1 : 12000	PROJ. NO.:	87X4660
CK'D BY:	EMH	DATE:	MAR. 29, 1994	FIG. NO.:	1

dose-response information.

Section 6.0 Risk Characterization; presents methods for evaluating information collected as part of the ecological assessment so that conclusions can be reached and risk management decisions supported.

Section 7.0 References for Sections 1.0 through 6.0.

2.0 AREA DESCRIPTION

The Cranston facility is located in the town of Cranston, Rhode Island, approximately ten miles south of Providence, Rhode Island. The site is located in the Pawtuxet River Basin. The Pawtuxet River drainage basin extends over an area of approximately 230 square miles. The Pawtuxet River flows through the facility on its way to Pawtuxet Cove, meandering through wooded areas, wetlands, commercial, industrial, and residential areas.

The Pawtuxet River, which separates the Production Area and Wastewater Treatment Area from the Warwick Area is the only surface water body located topographically downgradient of the site. Flow in the Pawtuxet River is regulated by two reservoir dams (Scituate Reservoir and Flat Rock Reservoir), the Pawtuxet Cove Dam, and multiple small mill dams throughout the length of the river. Land use in the watershed includes rural, urban, and industrial uses. Woodlands, wetlands, and grasslands exist in the reach of the river investigated. The Rhode Island State classification of water in the Pawtuxet River varies along the river, but is considered to be Class C/D downstream of the Cranston Sewage Treatment Plant; the facility reach is located within this area. Class C/D waters are suitable for migration of fish and have good aesthetic value but are not suitable for fishing or swimming.

The Pawtuxet River has received discharges from many industries in the past and present as well as from several sewage treatment plants. Before the industrial revolution (1800s) and dating back to the 1700s, forges and textile mills discharged to the Pawtuxet River; privies serving up to 3000 employees were positioned directly over the river. Currently, the waste water treatment plants of Warwick, West Warwick, and Cranston municipalities, as well as industrial metal plating operations and jewelry manufacturers, are upstream of the facility.

Water depth ranged from 2 to 9 feet along the facility reach during a bathymetric investigation conducted in July, 1990. Pools may have been caused by previous dredging activities or by erosional processes in the river. In general, shallow areas are colonized by aquatic macrophytes. These weed beds may simultaneously cause sediment depositions by a baffling effect and prevent erosion by stabilization of the sediment-water interface.

3.0 PROBLEM FORMULATION

Problem formulation provides a thorough description of potential ecological problems at each study site and defines objectives for the ecological assessment based on site information (USEPA, 1992a; Norton, et al., 1992). Problem formulation uses site descriptions from existing literature, any prior assessments, site history (including past, present, and future use), and physical features of each site to identify COCs, potential migration and exposure pathways, and potential ecological receptors (habitats and species) likely to be exposed.

3.1 Problem Formulation Results

This section describes determining surface water and sediment constituents of concern (COCs), primary and secondary exposure pathways, potential ecological receptors, the potential for adverse effects due to the presence of COCs, and appropriate assessment and measurement endpoints.

3.1.1 Site Description

The Facility site is divided into three areas: the Production Area, the Wastewater Treatment Area, and the Warwick Area. The first two areas are north of the Pawtuxet River. The Warwick Area is south of the river. Thirteen Soild Waste Management Units (SWMUs), two Areas of Concern (AOCs), and one Additional Area of Investigation (AAOI) have been identified as locations of former production facilities, waste treatment or waste storage sites, locations of documented spills, or areas of historical releases of hazardous substances. The Pawtuxet River has received discharges in the past, and continuing into the present, from a variety of industrial and municipal treatment plant sources. Currently, waste treatment plants for the municipalities of Warwick, West Warwick, and Cranston, as well as industrial metal plating operations and jewelry manufacturers, are upstream of the Facility. The physiochemical and biological characteristics of the Pawtuxet River were investigated between Elmwood Avenue and Rhodes-on-the-Pawtuxet, including the Facility reach (JT, 1992). Biological parameters (primarily fish and benthic invertebrate survey data) are comparable throughout the length investigated and indicate a moderately impacted system which is typical of a river flowing through commercial, industrial, and residential areas.

3.1.2 Constituents of Concern Selection

Chemical contaminants are the primary stressors evaluated in this report. Other anthropogenic, physical, or naturally-occurring stressors, as well as potential impacts to ecological receptors at the Facility from any significant offsite (non-Facility) stressor sources, were not investigated at this time. COCs are chemicals that were detected in Pawtuxet River sediments and which have the potential to adversely impact natural populations or ecosystems. Identification of detected chemical contaminants as COCs provides a focus for further investigation of potential threats to ecological receptors.

Analytical results from surface sediment and surface water are presented here as evidence of contamination within study areas. The following data were included as positive detections: data reported with a J or J equivalent qualifier (indicating an estimated concentration for tentatively identified compounds or when a result is less than the quantification limit but greater than zero), data reported with an E qualifier (indicating an estimated value because of the presence on interference), data reported with a B qualifier (compound was detected above the instrument detection limit but below the contract required detection

limit), data reported with a D qualifier (compound identified at a secondary dilution factor), data reported as < (less than), and data reported with a BW qualifier (indicating a post-digestion spike out of control limits with a value greater than the instrument detection limit but less than the contract required detection limit). Chemicals not detected (ND) at instrument detection limit were assigned a value of 0.5 of the detection limit if they were detected at least one time within a given reach. When no value was reported it was assumed that no analysis was performed for the chemical.

Environmental concentrations in sediments and surface water are represented by the geometric upper 95th confidence interval on the mean of a lognormal distribution. The intent of this approach is to estimate a Reasonable Maximum Exposure (RME) case (i.e., well above the average case) that is still within a range of possible exposures.

Initial screening for identification of analytes as COCs follows the path shown in Figure 3-1 and is based on the following criteria (USEPA, 1989a):

(a) ***Blank Contamination:*** As part of the data validation process, a chemical was not considered further if the maximum sample concentration did not exceed 10 times the highest blank for all common laboratory contaminants (acetone, 2-butanone, methylene chloride, toluene, and phthalates) or five times the highest blank for other chemicals.

(b) ***Frequency of Detection:*** Chemicals that are infrequently detected may be artifacts in the data due to sampling, analytical, or other problems. Chemicals were eliminated if they were detected in <5 percent of the samples.

(c) ***Essential Nutrients:*** Iron, magnesium, calcium, sodium, and potassium are considered essential macronutrients and are generally toxic only at very high concentrations. These constituents were eliminated as COCs.

(d) ***Water Chemistry:*** General water chemistry conditions (sulfates, carbonate, bicarbonate, chloride, TDS, and fluoride) are also generally toxic only at very high levels. These constituents were also eliminated as COCs.

Analytes that passed the initial screening were subjected to a final COC selection process using an additional suite of selection criteria:

(e) ***Upstream Background:*** If the concentration of an analyte was less than regulatory limits [(f) below], but its environmental concentration in the Facility and downstream reaches was >2x its concentration in the upstream (background) reach, it was identified as a COC, provided its physiochemical properties exceeded the threshold criteria for persistence [(g) below] and mobility [(h) below].

(f) ***Regulatory Limits:*** Applicable criteria and AWQC standards considered protective of most environmental receptors were EPA water quality criteria for the protection of aquatic life (USEPA, 1989c, 1992b) and NOAA sediment criteria (Long and Morgan, 1990). These criteria and standards represent maximum concentrations to which chronic exposure will not induce adverse effects. Analytes whose concentration in the Pawtuxet River exceeded these criteria were

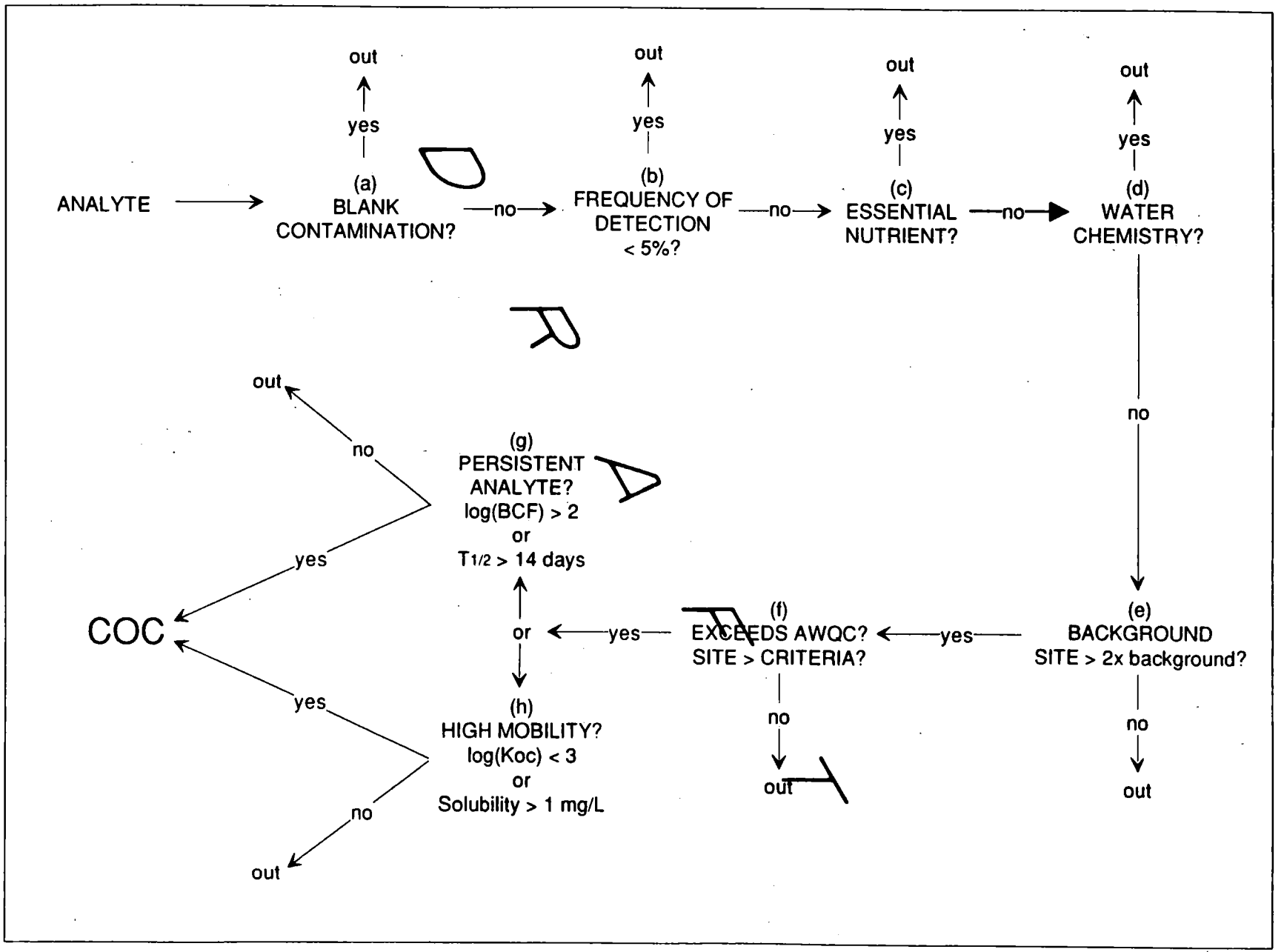


FIGURE 3-1 - Constituent of Concern Selection Process Flowchart

identified as COCs.

(g) ***Persistence:*** Persistence of constituents was estimated from the bioconcentration factor (BCF) and the degradation half-life ($t_{0.5}$) in soil (sediment). For aquatic ecosystems, analytes with bioconcentration factors (log(BCF)) <2 have a low potential for bioconcentration; log(BCF) values between 2 and 3 indicate a moderate potential; log(BCF) values >3 indicate a high potential. An analyte was identified as a COC if its log(BCF) was >2 or its half-life was >14 days (336 hours).

(h) ***Mobility:*** Mobility of constituents is a function of water solubility and the soil sorption constant (K_{oc}). Highly water soluble compounds tend to leach from wastes or contaminated soils and are generally mobile in both groundwater or surface water. Soil sorption indicates the tendency for a constituent to be adsorbed to soil or sediment particles. Constituents with log(K_{oc})values <2 are weakly sorbed (and thus more mobile); log(K_{oc}) values between 2 and 4 indicate moderate sorption; log(K_{oc}) values >4 indicate strong adsorption to soils. An analyte was classified as potentially mobile and as a COC if it exhibited a solubility >1 mg/L or a log(K_{oc}) < 3.

Physiochemical data for each analyte of interest to this assessment are provided in Table 3-1. If a constituent's environmental concentration exceeded a background and an AWQC and then its physiochemical properties suggested a moderate to high tendency for bioconcentration or mobility, it was selected as a COC. Analytes selected as COCs in sediment are shown in Table 3-2; those selected for surface water are shown in Table 3-3. Eight analytes were identified as surface water COCs, versus 54 sediment COCs. Bioassay testing was unable to identify significant mortality in test species exposed to surface water samples. This screening assessment will focus only on sediment COCs.

It is believed that many of the analytes selected as COCs were either not historically associated with Facility operations or have numerous other potential sources. Even though these COCs in the Pawtuxet River cannot be directly attributed to the Facility, they will be carried through the risk screening process to provide a more complete picture of ecotoxicological risks posed to the river ecosystem. At the conclusion of this screening assessment, a set of "indicator" COCs will be identified. These indicator COCs will represent classes of chemically and toxicologically similar compounds (metals, PAHs, etc.) that are: (a) associated with Facility operations, (b) likely to bioconcentrate or bioaccumulate through aquatic food webs, or (c) are the greatest contributors to total contaminant loading.

3.1.3 Exposure Pathway Identification

For exposures to occur, complete exposure pathways must exist; a complete pathway requires: (a) a source and mechanism for COC release, (b) a transport medium, (c) a point of environmental contact, and (d) an exposure route to the exposure point (USEPA, 1989a,b). If any of these four components is absent, a pathway is generally considered incomplete. However, the transport medium may be missing and the pathway still be complete if the contact point is directly at the contaminant release point. A generalized conceptual site model for potential migration and exposure pathways is presented in Figure 3-2.

With respect to the Pawtuxet River system, potential primary exposure pathways for aquatic receptors in the river, and terrestrial receptors that utilize the river, include: (a) direct contact with contaminated sediments or surface waters, (b) uptake through roots in contact with surface water or sediments, (c)

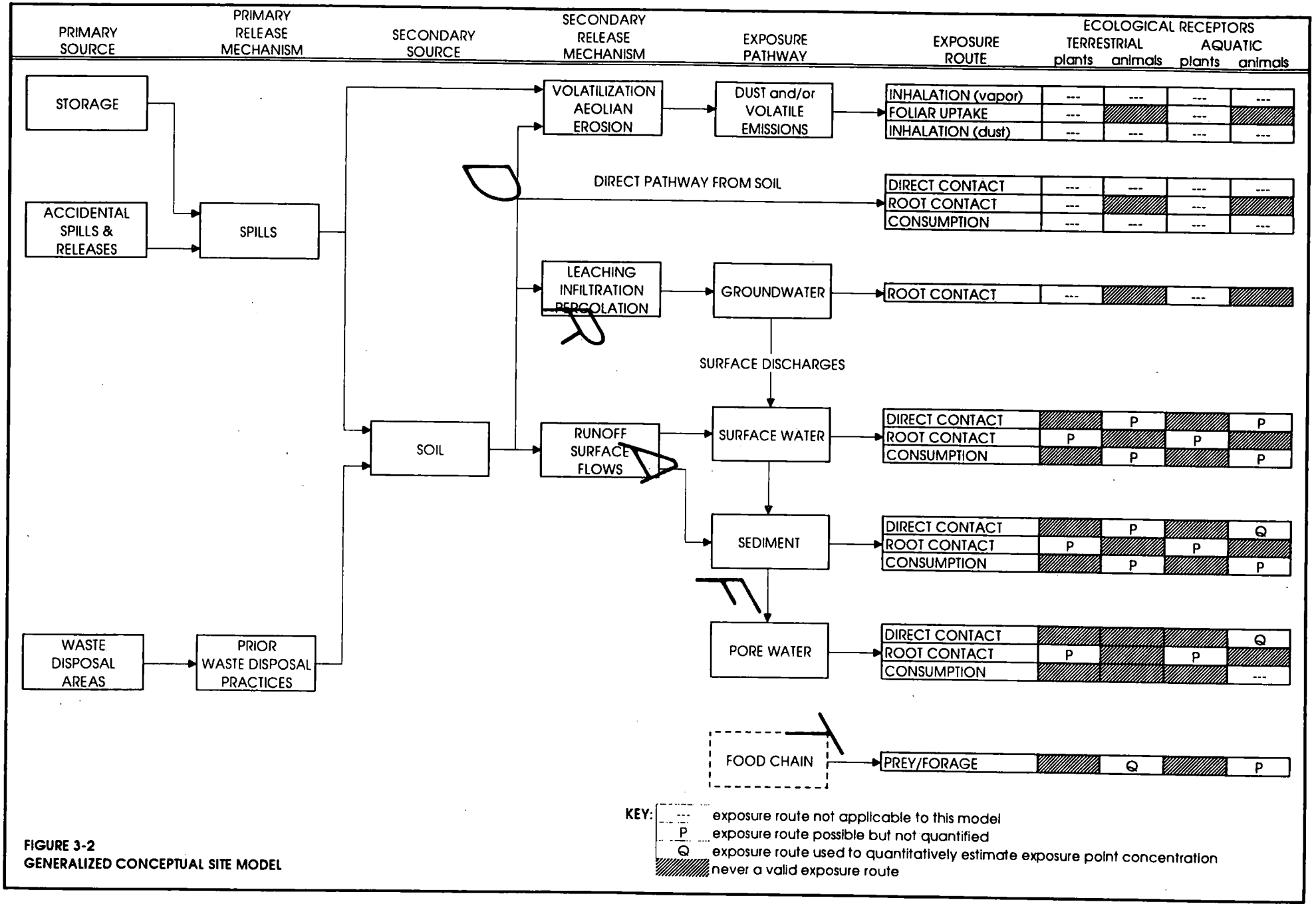


FIGURE 3-2
GENERALIZED CONCEPTUAL SITE MODEL

Table 3-1
CHEMICAL AND PHYSICAL PROPERTIES OF ANALYTES

Constituent of Concern	octanol-water partition coefficient log(Kow)	soil sorption coefficient log(Koc)[a]	water solubility[b] (mg/L)	surface water half-life[c] (days)	bioconcentration factor (log(BCF))[d]
ORGANICS					
1,2,4-trichlorobenzene	4.12	4.05	30	ND	2.26
1,1,2,2-tetrachloroethane	2.39	2.35	2900	45	0.90
1,2-dichlorobenzene	3.40	3.34	100	180	1.95
1,3-dichlorobenzene	3.60	3.54	123	180	1.81
1,4-dichlorobenzene	3.39	3.33	80	180	60.00
2-butanone	0.26	0.25	80	ND	-1.05
2-hexanone	1.38	1.36	ND	ND	0.82
2-methylphenol	1.95	1.92	ND	ND	1.25
4-chloroaniline	1.83	1.80	<1	ND	1.16
4-methyl-2-pentanone	ND	ND	ND	ND	ND
4-methylphenol	1.92	1.89	ND	<1	1.23
acetone	0.24	0.24	ND	7	-1.52
benzene	2.13	2.09	1780	16	0.81
chlorobenzene	2.84	2.79	490	150	1.01
chloroform	1.97	1.94	8220	180	0.78
ethylbenzene	3.15	3.10	152	10	1.83
iodomethane	3.15	3.10	10-50	ND	2.16
phenol	1.45	1.43	8400	2.4	0.3
Tinuvin 328	ND	ND	ND	ND	ND
toluene	2.69	2.64	<1	22	1.41
xylene (total)	3.04	2.99	0.30	28	1.85
PESTICIDES					
2,4-D	2.81	2.76	620	ND	1.49
4,4'-DDD	6.03	5.93	0.16	5694	4.24
4,4'-DDE	5.74	5.64	0.04	5694	5.04
4,4'-DDT	5.98	5.88	0.003	5694	4.58
aldrin	5.11	5.02	<1	584	3.65
BHC, alpha-	3.81	3.75	<1	135	2.67
BHC, beta-	3.89	3.82	<1	124	2.73
BHC, gamma-	3.72	3.66	8	240	2.40
Chlordane, alpha-	5.48	5.39	0	ND	4.15
Chlordane, gamma-	5.48	5.39	0	ND	4.15
dieldrin	4.09	4.02	<1	1095	2.88
dinoseb	ND	ND	<1	123	ND
disulfoton	3.94	3.87	<1	21	2.76
endosulfan II	3.55	3.49	ND	9.1	2.47
endrin	5.60	5.51	<1	ND	4.03
endrin aldehyde	5.60	5.51	ND	ND	4.03
heptachlor	5.00	4.92	<1	5	3.57
heptachlor epoxide	5.03	4.94	<1	552	3.59
methyl parathion	2.04	2.01	<1	365	1.32
pentachlorophenol	5.01	4.93	14	4.6	1.11
POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)					
2-methylnapthalene	3.86	3.79	<1	ND	2.70
acenaphthene	4.17	4.10	<1	102	2.94
acenaphthylene	3.92	3.85	<1	60	2.75
anthracene	4.40	4.33	<1	450	3.11
benzo(a)anthracene	5.60	5.51	<1	679	4.03
benzo(a)pyrene	6.31	6.20	<1	529	4.57

Table 3-1
CHEMICAL AND PHYSICAL PROPERTIES OF ANALYTES

Constituent of Concern	octanol-water partition coefficient log(Kow)	soil sorption coefficient log(Koc)[a]	water solubility[b] (mg/L)	surface water half-life[c] (days)	bioconcentration factor (log(BCF))[d]
benzo(b)fluoranthene	6.57	6.46	<1	610	4.76
benzo(g,h,i)perylene	7.23	7.11	<1	650	5.26
benzo(k)fluoranthene	6.84	6.72	<1	2139	4.97
chrysene	5.60	5.51	ND	993	4.03
dibenzo(a,h)anthracene	6.50	6.39	<1	942	4.71
dibenzofuran	4.12	4.05	<1	28	2.90
fluoranthene	5.33	5.24	<1	440	3.82
fluorene	4.18	4.11	ND	60	2.95
indeno(1,2,3-cd)pyrene	7.66	7.53	ND	730	5.59
naphthalene	3.36	3.30	<1	48	2.32
phenanthrene	4.46	4.38	<1	200	3.16
pyrene	5.18	5.09	<1	1898	3.71
POLYCHLORINATED BIPHENYLS (PCBs)					
PCB-1248	5.75	5.65	0.12	ND	4.14
PCB-1254	6.03	5.93	<1	ND	4.35
PHTHALATE ESTERS					
bis(2-ethylhexyl)phthalate	5.30	5.21	0.40	23	2.52
butylbenzylphthalate	4.05	3.98	13	7	2.82
di-n-butylphthalate	5.20	5.11	4500	23	1.95
di-n-octylphthalate	9.20	9.04	<0.1	28	6.76
dimethylphthalate	1.87	1.84	5	7	1.19
DIOXINS/FURANS					
2,3,7,8-TCDD	6.64	6.53	<1	591	4.82
INORGANICS		Kd			BCF
arsenic	NA	200	ND	NA	44
barium	NA	60	ND	NA	ND
chromium (III)	NA	850	ND	NA	16
cobalt	NA	45	ND	NA	ND
copper	NA	35	ND	NA	200
cyanide	NA	NA	ND	NA	0
lead	NA	900	ND	NA	49
manganese	NA	65	ND	NA	ND
mercury	NA	10	ND	NA	5500
nickel	NA	150	ND	NA	47
silver	NA	45	ND	NA	3080
tin	NA	45	ND	NA	0
vanadium	NA	1000.00	ND	NA	28[e]
zinc	NA	40.00	ND	NA	47

NA = not applicable; ND = no data available

[a] for organic compounds, log(Koc) = 0.00028 + (0.983 * log(Kow)) (EPA, 1993); for inorganics, Kd values from Baes et al., 1984

[b] Keith and Walters, 1992

[c] BEIA, 1990; Howard et al., 1991; HSDB, 1992

[d] log(BCF) = 0.76*log(Kow)-0.23 (Lyman et al., 1982); inorganic values are from EPA (1986, 1991)

[e] maximum value reported in Holdway et al., 1983

TABLE 3-2

Constituents of Concern for Sediment

ANALYTE	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS			Sediment Criteria[c]	Constituent of Concern[d]
	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	(mg/kg)	
INORGANICS											
Ammonia (as N)	3 /4	7.38E+00	**4.30E+02**	15 /16	3.71E+01	**1.40E+02**	7 /7	2.09E+01	**4.70E+01**	NA	No(b)
Antimony	1 /4	4.83E-01	**9.60E-01**	3 /16	6.95E-01	1.06E+00	ND	ND	ND	NA	No(b)
Arsenic	4 /4	8.74E+00	1.70E+01	16 /16	1.42E+01	3.61E+01	7 /7	5.04E+00	**1.05E+01**	NA	YES
Barium	4 /4	2.76E+01	8.24E+01	16 /16	1.35E+02	**3.80E+02**	7 /7	2.99E+01	**8.05E+01**	NA	YES
Beryllium	4 /4	7.14E-01	1.69E+00	16 /16	1.83E+00	3.28E+00	6 /7	5.59E-01	1.05E+00	NA	No(b)
Bicarbonate alkalinity	3 /4	1.77E+02	1.23E+03	11 /16	2.09E+02	4.92E+02	6 /7	1.06E+02	**1.55E+02**	NA	No(w)
Cadmium	2 /4	1.01E+00	**7.40E+00**	12 /16	6.59E+00	9.48E+00	3 /7	7.32E-01	2.14E+00	NA	No(b)
Calcium	4 /4	1.01E+03	2.22E+03	16 /16	2.30E+03	3.98E+03	7 /7	1.19E+03	3.35E+03	NA	No(b)
Chloride	3 /4	7.03E+01	**4.20E+02**	16 /16	1.70E+02	**6.80E+02**	5 /7	4.18E+01	1.49E+02	NA	No(b)
Chromium	4 /4	2.49E+01	**4.96E+01**	16 /16	2.61E+02	**1.26E+03**	7 /7	1.84E+01	4.16E+01	8.00E+01	YES
Cobalt	4 /4	3.39E+00	7.10E+00	16 /16	6.43E+00	8.17E+00	7 /7	3.09E+00	5.78E+00	NA	No(b)
Copper	4 /4	3.58E+01	**9.76E+01**	15 /16	3.00E+02	**1.08E+03**	6 /7	1.59E+01	3.99E+01	7.00E+01	YES
Cyanide	ND	ND	ND	9 /16	7.14E+00	1.17E+01	ND	ND	ND	NA	YES
Iron	4 /4	8.05E+03	1.45E+04	16 /16	1.52E+04	1.90E+04	7 /7	7.87E+03	9.59E+03	NA	No(b)
Lead	4 /4	4.71E+01	**1.73E+02**	16 /16	1.98E+02	**8.29E+02**	7 /7	4.90E+01	**1.28E+02**	3.50E+01	YES
Magnesium	4 /4	1.32E+03	2.15E+03	16 /16	2.02E+03	2.83E+03	7 /7	9.89E+02	1.29E+03	NA	No(e)
Manganese	4 /4	1.57E+02	3.07E+02	16 /16	2.83E+02	3.64E+02	7 /7	1.78E+02	2.89E+02	NA	No(b)
Mercury	1 /4	5.12E-02	1.33E-01	10 /16	5.31E-01	**2.80E+00**	1 /7	3.81E-02	6.15E-02	1.50E-01	YES
Nickel	3 /4	8.62E+00	2.95E+01	12 /16	3.63E+01	1.48E+02	4 /7	5.80E+00	1.23E+01	3.00E+01	YES
Nitrate/Nitrite (as N)	3 /4	2.57E+00	**5.90E+01**	10 /16	1.50E+01	**1.60E+02**	5 /7	1.80E+00	1.01E+01	NA	No(w)
Orthophosphate	4 /4	3.86E+00	**1.10E+01**	15 /16	2.31E+01	**1.30E+02**	7 /7	1.46E+01	**3.40E+01**	NA	No(w)
Potassium	4 /4	5.31E+02	1.20E+03	15 /16	1.11E+03	1.40E+03	5 /7	3.75E+02	6.27E+02	NA	No(b)
Selenium	2 /4	3.97E-01	8.85E-01	8 /16	6.28E-01	7.82E-01	3 /7	3.34E-01	4.47E-01	NA	No(b)
Silver	ND	ND	ND	6 /16	1.29E+00	2.14E+00	ND	ND	ND	1.00E+00	YES
Sodium	2 /4	1.54E+02	3.70E+02	10 /16	3.53E+02	4.84E+02	4 /7	1.14E+02	2.23E+02	NA	No(e)
Sulfate	2 /4	3.67E+01	**1.90E+02**	13 /16	5.65E+02	**2.50E+03**	5 /7	6.29E+01	1.92E+02	NA	No(w)
Sulfide	3 /4	1.18E+02	1.34E+03	8 /16	1.30E+03	1.32E+04	3 /7	3.36E+01	7.33E+01	NA	No(w)
Thallium	2 /4	3.97E-01	8.85E-01	7 /16	5.14E-01	6.73E-01	2 /7	3.03E-01	4.00E-01	NA	No(b)
Tin	ND	ND	ND	4 /16	1.06E+01	1.58E+01	ND	ND	ND	NA	YES
Total alkalinity	3 /4	1.77E+02	1.23E+03	11 /16	2.09E+02	4.92E+02	6 /7	1.06E+02	**1.55E+02**	NA	No(w)
Total organic carbon	2 /2	SS	**1.10E+04**	8 /8	2.93E+04	**8.50E+04**	3 /3	1.27E+03	**3.70E+03**	NA	No(w)
Vanadium	4 /4	6.85E+00	**1.45E+01**	16 /16	1.84E+01	**4.94E+01**	5 /7	3.42E+00	7.00E+00	NA	YES
Zinc	4 /4	7.43E+01	**2.29E+02**	16 /16	2.11E+03	**1.39E+04**	7 /7	7.55E+01	1.66E+02	1.20E+02	YES
ORGANICS											
1,1,2,2-Tetrachloroethane	ND	ND	ND	1 /16	2.07E-01	**3.40E-01**	1 /7	6.36E-02	**5.20E-02**	NA	No(x)
1,2,4-Trichlorobenzene	ND	ND	ND	1 /15	1.04E+00	**4.00E-01**	ND	ND	ND	NA	No(z)

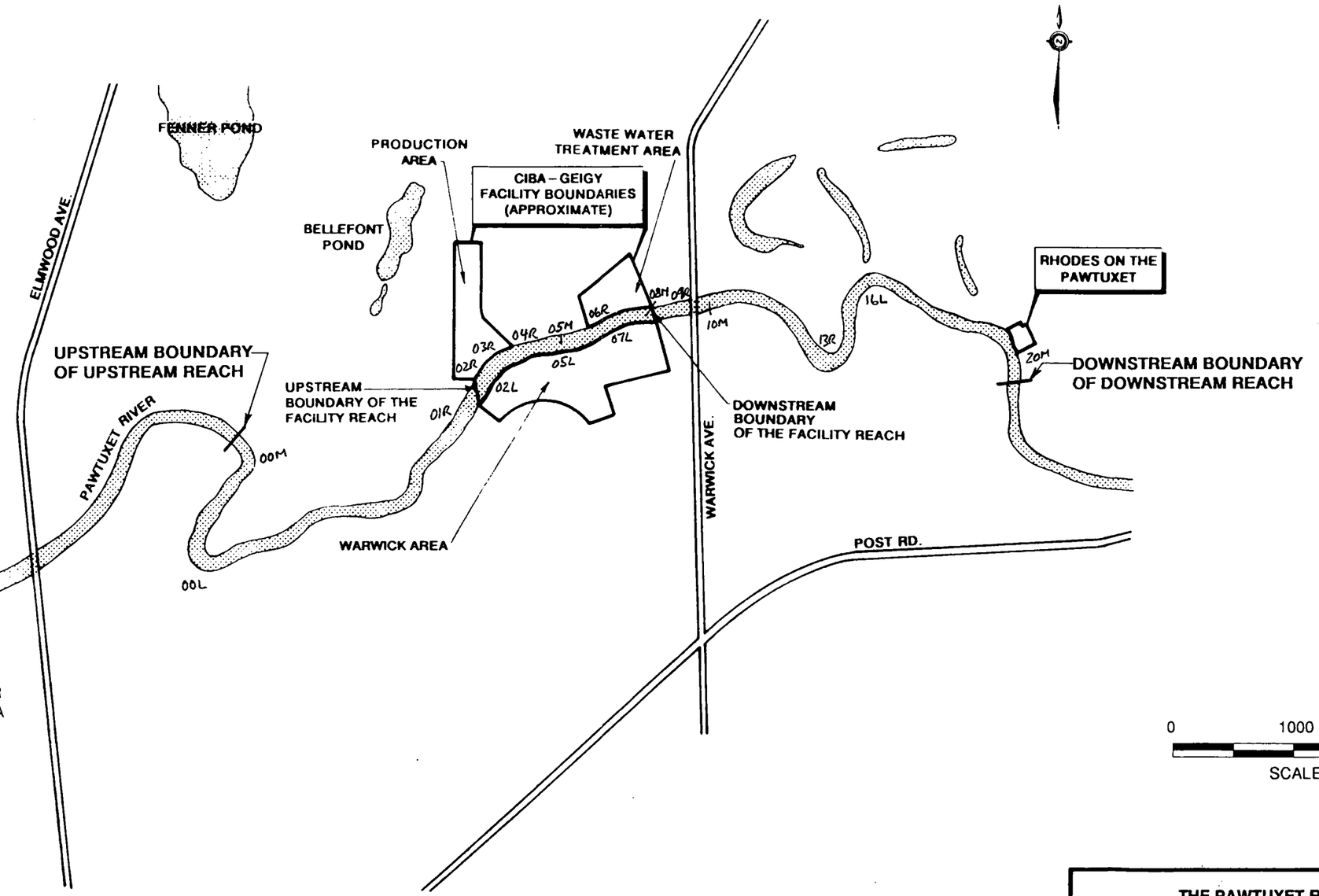
TABLE 3-2

Constituents of Concern for Sediment

ANALYTE	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS			Sediment Criteria[c] (mg/kg)	Constituent of Concern[d]
	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)		
1,2-Dichlorobenzene	1 /4	5.05E-01	1.20E-01	6 /15	6.90E-01	1.01E+00	ND	ND	ND	NA	YES
1,3-Dichlorobenzene	ND	ND	ND	1 /15	1.08E+00	**6.90E-01**	ND	ND	ND	NA	No(z)
1,4-Dichlorobenzene	ND	ND	ND	3 /15	9.48E-01	1.46E+00	ND	ND	ND	NA	YES
2-Butanone	1 /4	1.90E-01	4.58E-01	2 /16	4.14E-01	**7.40E-01**	3 /7	1.45E-01	1.71E-01	NA	No(c)
2-Hexanone	1 /4	1.54E-01	2.28E-01	1 /16	3.94E-01	**3.80E-01**	ND	ND	ND	NA	No(c)
4-Chloroaniline	ND	ND	ND	2 /16	1.34E+00	2.17E+00	ND	ND	ND	NA	No(x)
4-Methyl-2-pentanone	ND	ND	ND	1 /16	3.85E-01	**2.60E-01**	ND	ND	ND	NA	No(c)
2-Methylphenol	ND	ND	ND	1 /15	1.03E+00	1.32E+00	ND	ND	ND	NA	No(x)
4-Methylphenol	1 /4	7.42E-01	1.09E+00	3 /15	8.85E-01	1.26E+00	1 /6	5.15E-01	**1.70E-01**	NA	No(b)
Acetone	ND	ND	ND	1 /16	4.08E-01	**6.40E-01**	ND	ND	ND	NA	No(c)
Benzene	ND	ND	ND	1 /16	1.89E-01	**8.60E-02**	ND	ND	ND	NA	No(x)
Chlorobenzene	1 /4	1.40E-01	5.11E-01	7 /16	5.02E-01	2.12E+00	2 /7	6.86E-02	8.11E-02	NA	YES
Ethylbenzene	ND	ND	ND	1 /16	1.85E-01	**6.10E-02**	ND	ND	ND	NA	No(x)
Phenol	ND	ND	ND	ND	ND	ND	1 /7	6.95E-01	8.31E-01	NA	No(b)
Tinuvin 328	ND	ND	ND	5 /15	5.55E+00	1.68E+01	2 /6	1.74E+00	**1.20E+00**	NA	YES
Toluene	1 /4	7.21E-02	9.35E-02	10 /15	8.89E+01	**8.60E+02**	2 /7	7.04E-02	8.87E-02	NA	YES
Xylene, m- & p-	1 /4	5.97E-02	**2.10E-02**	2 /16	2.28E-01	4.93E-01	ND	ND	ND	NA	YES
Xylene, o-	1 /4	8.07E-02	**7.00E-02**	2 /16	1.91E-01	**2.30E-01**	ND	ND	ND	NA	YES
PESTICIDES											
2,4-D	ND	ND	ND	2 /16	1.34E-01	2.02E-01	ND	ND	ND	NA	YES
4,4'-DDD	1 /4	3.47E-03	**2.80E-02**	1 /16	1.42E-02	**3.00E-02**	1 /6	1.74E-03	**1.90E-03**	2.00E-03	No(b)
4,4'-DDE	ND	ND	ND	3 /16	1.89E-02	5.43E-02	1 /6	2.10E-03	6.28E-03	2.00E-03	YES
4,4'-DDT	1 /4	4.12E-03	**7.00E-03**	5 /16	2.84E-02	8.15E-02	1 /6	3.47E-03	**6.20E-03**	1.00E-03	YES
Aldrin	2 /4	5.15E-03	**4.90E-02**	3 /16	1.76E-02	**3.50E-02**	4 /6	6.73E-03	1.14E-02	NA	No(b)
BHC, alpha-	ND	ND	ND	2 /16	1.29E-02	3.79E-02	ND	ND	ND	NA	YES
BHC, gamma-	2 /4	5.55E-03	**2.90E-02**	3 /16	1.82E-02	**2.10E-02**	2 /6	2.77E-03	8.19E-03	NA	No(b)
Chlordane, alpha-	1 /4	3.11E-03	**1.80E-02**	1 /16	1.37E-02	**1.80E-02**	1 /6	2.10E-03	6.28E-03	5.00E-04	No(b)
Chlordane, gamma-	2 /4	3.68E-03	**2.30E-02**	4 /16	1.87E-02	5.09E-02	1 /6	1.97E-03	**4.10E-03**	5.00E-04	YES
Dieldrin	ND	ND	ND	5 /16	1.55E-02	4.51E-02	2 /6	2.47E-03	9.89E-03	2.00E-05	YES
Dinoseb	ND	ND	ND	1 /16	1.71E-02	**4.90E-03**	2 /7	1.12E-02	1.36E-02	NA	No(z)
Disulfoton	ND	ND	ND	3 /16	9.76E-02	**1.20E-01**	1 /6	4.80E-03	1.83E-02	NA	YES
Endosulfan II	ND	ND	ND	1 /16	2.83E-02	**1.50E-03**	ND	ND	ND	NA	No(z)
Endrin	ND	ND	ND	2 /16	1.39E-02	**1.00E-02**	ND	ND	ND	2.00E-05	YES
Endrin aldehyde	ND	ND	ND	ND	ND	ND	1 /6	4.80E-03	1.83E-02	NA	No(b)
Heptachlor	1 /4	2.87E-03	**1.30E-02**	4 /16	1.82E-02	**4.40E-02**	4 /6	4.61E-03	1.16E-02	NA	YES
Heptachlor epoxide	ND	ND	ND	1 /16	1.36E-02	4.04E-02	1 /6	2.47E-03	9.92E-03	NA	No(z)
Methyl parathion	ND	ND	ND	3 /15	1.77E-02	2.72E-02	1 /7	1.01E-02	**5.10E-03**	NA	No(x)

TABLE 3-2
Constituents of Concern for Sediment

ANALYTE	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS			Sediment Criteria[c] (mg/kg)	Constituent of Concern[d]
	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)		
Pentachlorophenol	ND	ND	ND	1 /15	5.06E+00	6.38E+00	ND	ND	ND	NA	No(z)
PAHs											
2-Methylnaphthalene	1 /4	1.52E+00	7.20E+00	5 /15	8.95E-01	1.19E+00	1 /6	4.30E-01	**4.90E-02**	6.50E-02	No(b)
Acenaphthene	1 /4	5.73E-01	**2.00E-01**	6 /15	4.93E-01	**3.60E-01**	ND	ND	ND	1.50E-01	No(b)
Acenaphthylene	ND	ND	ND	3 /15	6.77E-01	**1.10E-01**	ND	ND	ND	NA	YES
Anthracene	4 /4	1.73E-01	2.85E-01	14 /15	1.23E+00	**9.30E-01**	6 /6	9.71E-02	1.60E-01	8.50E-02	YES
Benzo(a)anthracene	4 /4	6.95E-01	**1.70E+00**	14 /15	2.35E+00	**3.80E+00**	6 /6	4.77E-01	7.20E-01	2.30E-01	YES
Benzo(a)pyrene	3 /4	8.45E-01	1.79E+00	11 /15	2.43E+00	**4.10E+00**	5 /6	5.27E-01	7.23E-01	4.00E-01	YES
Benzo(b)fluoranthene	4 /4	1.20E+00	**3.30E+00**	14 /15	3.83E+00	**8.90E+00**	6 /6	7.51E-01	1.25E+00	NA	YES
Benzo(g,h,i)perylene	3 /4	8.47E-01	1.96E+00	10 /15	2.51E+00	**4.80E+00**	3 /6	5.54E-01	7.95E-01	NA	YES
Benzo(k)fluoranthene	4 /4	1.30E+00	**3.70E+00**	14 /15	4.08E+00	**9.30E+00**	6 /6	8.46E-01	1.43E+00	NA	YES
Chrysene	4 /4	9.14E-01	**2.40E+00**	14 /15	3.17E+00	**6.90E+00**	7 /7	6.23E-01	**1.40E+00**	4.00E-01	YES
Dibenz(a,h)anthracene	2 /4	4.86E-01	**7.00E-01**	5 /15	8.74E-01	1.39E+00	2 /6	4.10E-01	**2.00E-01**	6.00E-02	No(b)
Dibenzofuran	1 /4	5.05E-01	**1.20E-01**	7 /15	4.20E-01	**3.20E-01**	1 /6	4.21E-01	**6.70E-02**	NA	YES
Fluoranthene	4 /4	1.97E+00	5.82E+00	15 /15	4.65E+00	**1.40E+01**	7 /7	1.38E+00	**3.60E+00**	6.00E-01	YES
Fluorene	4 /4	1.04E-01	**1.80E-01**	10 /15	1.85E+00	**7.90E-01**	2 /6	3.07E-01	**1.00E-01**	3.50E-02	YES
Indeno(1,2,3-cd)pyrene	3 /4	7.79E-01	**1.60E+00**	11 /15	2.48E+00	**4.30E+00**	4 /6	4.56E-01	6.84E-01	NA	YES
Naphthalene	1 /4	3.91E-01	**9.30E-02**	6 /15	1.28E+00	2.95E+00	3 /6	2.54E-01	**2.30E-01**	3.40E-01	YES
Phenanthrene	4 /4	1.00E+00	**1.60E+00**	15 /15	1.97E+00	5.49E+00	7 /7	6.01E-01	8.62E-01	2.25E-01	YES
Pyrene	4 /4	1.19E+00	**2.30E+00**	15 /15	2.89E+00	7.72E+00	7 /7	7.76E-01	1.12E+00	3.50E-01	YES
PCBs											
PCB-1248	ND	ND	ND	2 /16	1.61E-01	5.96E-01	ND	ND	ND	5.00E-02	YES
PCB-1254	ND	ND	ND	3 /16	3.77E-01	1.49E+00	ND	ND	ND	5.00E-02	YES
PHTHALATE ESTERS											
Bis(2-ethylhexyl)phthalate	1 /4	1.08E+00	2.48E+00	7 /16	2.29E+00	5.21E+00	3 /7	1.00E+00	1.68E+00	NA	No(c)
Butylbenzylphthalate	ND	ND	ND	1 /15	1.05E+00	**2.20E-01**	1 /6	5.45E-01	**2.40E-01**	NA	No(z)
Di-n-butylphthalate	1 /4	3.80E-01	**3.30E-02**	5 /16	6.22E-01	1.45E+00	1 /6	4.24E-01	**5.30E-02**	NA	YES
Di-n-octylphthalate	ND	ND	ND	2 /15	1.06E+00	1.41E+00	1 /6	7.31E-01	9.77E-01	NA	YES
Dimethylphthalate	ND	ND	ND	1 /13	1.47E+00	3.33E+00	ND	ND	ND	NA	No(x)
DIOXINS/FURANS											
DCDF	ND	ND	ND	1 /15	2.03E+00	5.00E-01	ND	ND	ND	NA	No(z)
HxCDD	ND	ND	ND	4 /16	1.05E-03	4.10E-03	ND	ND	ND	NA	YES
HxCDF	ND	ND	ND	4 /16	6.90E-04	3.27E-03	ND	ND	ND	NA	YES
PeCDD	ND	ND	ND	2 /16	1.79E-04	3.62E-04	ND	ND	ND	NA	YES
PeCDF	ND	ND	ND	5 /16	4.02E-04	1.31E-03	ND	ND	ND	NA	YES
TCDD	ND	ND	ND	3 /16	1.76E-04	3.18E-04	ND	ND	ND	NA	YES
TCDF	ND	ND	ND	4 /16	1.63E-04	3.11E-04	1 /7	5.95E-05	7.67E-05	NA	YES



LEGEND

MAJOR SURFACE WATER BODIES IN MAPPED AREA

BASE MAP SOURCE:

AERIAL PHOTOGRAPHS BY GEOD CORPORATION OF NEWFOUNDLAND, NEW JERSEY.
DATE FLOWN: 2 APRIL 1989.

THE PAWTUXET RIVER REACHES: UPSTREAM, FACILITY, AND DOWNSTREAM

WOODWARD – CLYDE CONSULTANTS

CONSULTING ENGINEERS, GEOLOGISTS AND ENVIRONMENTAL SCIENTISTS
WAYNE, NEW JERSEY

DR. BY: KJFH	SCALE: 1 : 12000	PROJ. NO.: 87X4660
CK'D BY: EMH	DATE: MAR. 29, 1994	FIG. NO.: 1

TABLE 3-2
Constituents of Concern for Sediment

ANALYTE	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS			Sediment Criteria[c] (mg/kg)	Constituent of Concern[d]
	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)	Frequency of Detection	Mean[a] (mg/kg)	95% CI on Mean[b] (mg/kg)		
TRCDF	1/4	1.18E+00	4.30E-01	ND	ND	ND	1/6	1.00E+00	**2.60E-01**	NA	No(b)

NOTES

[a] If the distribution is normal or if the distribution is log-normal, the number of detects >7, and frequency of detection >50%, an arithmetic mean is given. If the distribution is log-normal and either the number of detects < 7 or the frequency of detection is < 50%, a geometric mean is given.

[b] If the distribution is normal or if the distribution is log-normal, the number of detects >7, and the frequency of detection >50%, an arithmetic upper 95% confidence interval (CI) on the mean is given. If the distribution is log-normal and either the number of detects < 7 or the frequency of detection is < 50%, a geometric upper 95% CI on the mean is given.

[c] Long & Morgan, 1990 (ER-L value)

[d] Constituent of Concern selection criteria:

- YES sediment COC
- No(b) facility concentration < 2x upstream concentration
- No(w) water chemistry
- No(s) facility sediment concentration < sediment criteria
- No(e) essential nutrient
- No(c) common laboratory contaminant
- No(x) log(BCF) < 2
- No(z) frequency of detection <5%

ND = compound not detected

SS = not applicable if sample size <= 2.

bold = If the upper 95% CI on the mean exceeds the maximum detected concentration or if the sample size <= 2, the maximum detected concentration is substituted.

UPSTREAM STATION SAMPLE NUMBERS = SD-00I*IB-2, SD-00M, SD-01R, SD-01R*IB-2

FACILITY STATION SAMPLE NUMBERS = SD-02L, SD-02L*IB-2, SD-02R, SD-02R*IB-2, SD-03L*IB-2, SD-03R*IB-2, SD-04R*IB-2, SD-05L, SD-05M*IB-2, SD-06L, SD-07L*IB-2, SD-07R, SD-08M*IB-2, SD-08R, SD-F03L, SD-F03R

DOWNSTREAM STATION SAMPLE NUMBERS = SD-09R*IB-2, SD-10M, SD-13R*IB-2, SD-16M*IB-2, SD-20M, SD-20M*IB-2, SD-09A

Table 3-3
Constituents of Concern for Surface Water

ANALYTE	UPSTREAM STATIC			FACILITY & DOWNSTREAM STATIONS			AWQC[b] (mg/L)	Constituent of Concern[c]
	Frequency of Detection	Mean Concentration (mg/L)	Upper 95% CI Concentration[a] (mg/L)	Frequency of Detection	Mean Concentration (mg/L)	Upper 95% CI Concentration[a] (mg/L)		
INORGANICS								
Ammonia	4/4	1.11E+00	2.10E+00	8/8	1.38E+00	1.90E+00	NA	No(b)
Barium	4/4	1.47E+01	1.59E+01	6/6	1.51E+01	1.60E+01	1.09E-01	No(b)
Cyanide	2/4	8.50E-03	1.83E-02	4/6	9.71E-03	1.54E-02	5.20E-03	No(b)
Iron	4/4	4.75E+02	5.90E+02	6/6	5.13E+02	6.14E+02	1.00E+00	No(b)
Lead	4/4	6.26E+00	1.76E+01	5/6	4.09E+00	6.23E+00	3.20E-03	No(b)
Magnesium	4/4	1.38E+03	1.63E+03	6/6	1.48E+03	1.72E+03	1.60E-03	No(b)
Manganese	4/4	9.97E+01	1.40E+02	6/6	1.13E+02	1.40E+02	1.10E-02	No(b)
Nickel	1/4	1.27E+01	2.23E+01	2/6	1.38E+01	2.09E+01	1.60E-01	No(b)
Orthophospahte	4/4	2.74E-01	6.60E-01	8/8	2.92E-01	4.60E-01	NA	No(b)
Potassium	ND	ND	ND	4/6	2.54E+03	3.48E+03	1.30E-04	No(e)
Silver	ND	ND	ND	1/6	6.98E+00	1.37E+01	3.90E-04	YES
Sodium	4/4	2.33E+04	3.26E+04	6/6	2.65E+04	3.34E+04	4.80E-01	No(b)
Sulfate	4/4	1.70E+01	2.40E+01	8/8	2.01E+01	2.60E+01	NA	No(b)
Sulfide	2/4	2.16E+00	1.30E+01	6/8	1.89E+00	5.29E+00	NA	No(b)
ORGANICS								
Chlorobenzene	2/4	1.58E+00	1.00E+00	5/8	1.53E+00	1.20E+00	1.16E-02	No(b)
Chloroform	ND	ND	ND	2/8	2.58E+00	2.94E+00	1.40E-02	YES
Iodomethane	ND	ND	ND	1/8	2.43E+00	2.00E+00	NA	No(x)
toluene	1/4	2.12E+00	1.30E+00	6/8	1.65E+00	2.00E+00	1.04E-02	No(b)
xylene (m & p)	1/4	2.12E+00	1.30E+00	4/8	1.82E+00	1.60E+00	1.00E-03	No(b)
xylene (o)	1/4	1.67E+00	1.80E+00	3/8	1.43E+00	2.30E+00	1.00E-03	No(b)
PESTICIDES								
4,4'-DDE	ND	ND	ND	2/8	5.32E-03	6.11E-03	1.05E+00	No(s)
4,4'-DDT	ND	ND	ND	1/8	1.10E-02	1.40E-02	1.00E-06	YES
beta-BHC	ND	ND	ND	1/8	6.49E-03	1.14E-02	2.40E-04	YES
Dieldrin	ND	ND	ND	1/8	5.11E-03	5.72E-03	1.90E-06	YES
Dimethoate	ND	ND	ND	1/7	3.57E-01	6.40E-02	NA	YES
Disulfoton	ND	ND	ND	2/7	1.96E-01	2.20E-02	NA	YES
Methyl parathion	1/4	5.12E-02	2.00E-02	2/7	6.92E-02	1.05E-01	1.30E-05	YES

Table 3-3
Constituents of Concern for Surface Water

NOTES

[a] Upper 95% confidence interval concentration = environmental concentration

[b] chronic NAWQ value given in Suter et al., 1992

[c] Constituent of Concern Selection

YES: COC in sediment
No(b): facility/downstream concentration is less than 2 x upstream concentration
No(f): frequency of detection < 5%
No(c): common laboratory contaminant
No(e): essential macronutrient
No(w): water chemistry
No(x): log(BCF) < 2
No(s): facility surface water concentration < AWQC

NA = not available

ND = not detected

UPSTREAM STATIONS = 00, 01

FACILITY & DOWNSTREAM STATIONS = 02, 03, 04, 05, 06, 07, 08, 09, 10, 13, 16, 20

ingestion of contaminated surface waters, (d) incidental ingestion of contaminated sediments by either aquatic or terrestrial consumers, and (e) secondary exposure pathways for both aquatic and terrestrial receptors that involve ingestion of contaminants which have bioaccumulated into forage or prey items. Specific exposure pathways include:

(a) ***Direct Contact (surface water/sediment):*** Aquatic organisms inhabiting contaminated waters were assumed to be in equilibrium with contaminants in surface water; however, because COCs were assumed to enter surface waters from pore waters, pore waters were taken as the exposure point for aquatic biota. Terrestrial organisms may come in contact with water-borne contaminants as a result of wading or swimming in contaminated waters. However, significant exposure via dermal contact would be limited to organic contaminants which are lipophilic and can transit epidermal barriers; this is seen as an unlikely exposure pathway for adult mammals or birds.

(b) ***Root Contact (surface water/sediment):*** Contaminants may be taken-up by either terrestrial plants or aquatic macrophytes whose roots are in contact with sediment or surface waters. Contaminants may be translocated into edible foliage or reproductive structures (seeds). Plants were assumed to be exposed primarily through contact with contaminated sediments rather than with contaminated surface water.

(c) ***Surface Water Consumption:*** Terrestrial receptors may ingest water-borne contaminants if impacted surface waters are used as a drinking water source. Aquatic organisms inhabiting contaminated waters were assumed to be in equilibrium with contaminants in surface water.

(d) ***Consumption (sediment):*** Some aquatic organisms consume sediment and ingest organic material from the sediment. Inadvertent ingestion of sediments may occur when either terrestrial or aquatic consumers ingest benthic organisms or plant materials.

(e) ***Food Web Interactions:*** Indirect exposure pathways involve contaminants that biomagnify within the food chain. Contaminants bound to soil or sediment are assumed to be bioavailable only after they partition into the water phase. Water-borne contaminants may bioaccumulate into plant tissues in contact with soil, sediment, or surface water or into terrestrial or aquatic species ingesting soil, sediment, or surface water. As these plants and/or animals are consumed, contaminants may be be passed up the food chain to impact organisms within higher trophic levels.

3.1.4 Ecological Receptor Identification

In March, 1992, a terrestrial/riparian reconnaissance, a fish population survey, and a benthic invertebrate survey were conducted at the Facility (IT, 1992). These investigations included three localities: the Facility itself, the region east and downstream of the Facility boundary just west of the Warwick Avenue bridge to Rhodes-on-the Pawtuxet, and the region west and upstream of the Facility boundary near Atlantic Rubber and Tubing to the Elmwood Avenue bridge. Each locality included upland areas and riparian zones bordering the Pawtuxet River; the downstream locality included a wetland area.

Aquatic Species: Fish populations were sampled through use of a boat-mounted electroshocker and gill nets. Sampling was conducted from areas upstream of the I-95 bridge down to Rhodes-on-the-Pawtuxet and the following species were collected (the total number of fish collected of that species are contained in

parentheses): White sucker - *Catastomus commersoni* (298), Common carp - *Cyprinus carpio* (53), Golden shiner - *Notemigonus crysoleucas* (9), Black bullhead - *Ameirus melas* (5), Bluegill - *Lepomis macrochirus* (4), Pumpkinseed - *Lepomis gibbosus* (2), Redear sunfish - *Lepomis microlophus* (1), and American eel - *Anguilla rostrata* (1).

White suckers were numerically dominant at all areas surveyed. Common carp were abundant, particularly near the Production Area end of the facility. Golden shiner were common. All other species collected were relatively few in number. A benthic invertebrate survey was conducted in June, 1993. Various species of aquatic insect larvae were identified, as well as leeches, snails, and flatworms. Tubifex worms were the numerically dominant species in the majority of samples. Amphibians were represented by toads (*Bufo americanus*). The resident species are generally considered tolerant of chemical and physical disturbances.

Terrestrial Species: The terrestrial survey identified twenty-eight species of upland plants and twenty-six species of riparian/wetland plants at and near the Facility. Twenty-six species of birds were identified as well. These included the great blue heron (*Ardea herodias*), mallard duck (*Anas platyrhyncus*), and red-tailed hawk (*Buteo jamaicensis*). Five mammal species were identified, including the Eastern gray squirrel (*Sciurus carolinensis*) and the raccoon (*Procyon lotor*).

3.1.5 Potential Adverse Effects

For adverse ecological effects to be possible, a study site must: (a) contain COCs in abiotic media at detectable and biologically significant concentrations, (b) provide exposure pathways linking contaminants to receptors, and (c) have ecological receptors that either utilize the site, are present nearby, or are in range of COCs migrating from the site. If these three fundamental conditions cannot be met, the probability of adverse effects due to site-related contaminants is minimal.

It has been shown that the Pawtuxet River at and below the Facility contains COCs in abiotic media at detectable concentrations and provides exposure pathways linking these COCs to both onsite and offsite ecological receptors. These receptors could have experienced, or could be experiencing, acute or chronic toxic effects due to exposure to site-specific COCs.

3.1.6 Endpoints

An ecological assessment must define site-specific assessment endpoints, with associated measurement endpoints. An assessment endpoint is a formal expression of the actual environmental values that are to be protected; a measurement endpoint is a measurable ecological characteristic that is relatable to the valued environmental characteristic chosen as an assessment endpoint (USEPA, 1989a, 1992a; Suter, 1993). Assessment endpoints were based on potential effects at the population level of biological organization, as these are usually better defined and more predictable with current data and methods than are responses at these higher levels of biological organization (USEPA, 1989a). Toxic effects due to COCs may take the form of reduced reproductive success in individual organisms and such potential adverse effects could lead directly to a reduction in total population abundance for site-specific ecological receptors. Measurement endpoints were published results of laboratory or field toxicity tests performed on

fish, mammal, and avian species that share an operational relationship with previously defined assessment endpoints; they serve as surrogates for the assessment endpoints (Suter, 1993). Endpoints that may be appropriate for this phase of the Pawtuxet River assessment process are summarized in Table 3-4.

3.2 Problem Formulation Summary

It was determined that COCs extant in the Facility reach could be contributing to the potential for adverse effects in ecological receptors in the Pawtuxet River. Fish and invertebrate species are directly exposed to COCs in surface water and sediments, while higher trophic level receptors (e.g., great blue heron, raccoon) may be exposed to COCs bioconcentrated in their prey species (fish and invertebrates).

Principal questions of interest to this screening assessment are:

- Are ecological receptors currently exposed to site-related COCs at levels capable of causing harm?
- If adverse ecological effects are observed or predicted, what are the types, extent, and severity of effects?
- To what extent do contaminants present in the upstream reach contribute to the potential for any adverse impacts within the Facility reach?
- Are there contaminants whose potential for adverse impacts is confined to the Facility reach?
- To what extent do contaminants present in the Facility reach contribute to the potential for adverse impacts within the downstream reach?

TABLE 3-4
RELATIONSHIP OF ENDPOINTS

ASSESSMENT GOAL	ASSESSMENT ENDPOINT	INDICATORS OF EFFECTS	MEASUREMENT ENDPOINTS
Minimal impacts to aquatic species; primarily aquatic vertebrates	(a) No probability for a reduction of >10% in population abundance of fish or invertebrate species [a]	(1) laboratory toxicity to common fish test species (2) laboratory toxicity to common invertebrate test species (3) species-specific field or laboratory toxicity data (4) benthic community parameters with respect to a reference location (5) sediment bioassay tests (6) surface water bioassay tests (7) pore water bioassay tests	fish NOEL aquatic invertebrate NOEL community indicies species richness (S) species diversity (H') species dominance (D) reduced survivorship in laboratory tests or in comparison to a "reference" area
Minimal impacts to piscivorous terrestrial wildlife and avian species	(b) No probability for a reduction of >10% in population abundance of piscivorous wildlife or avian species	(1) laboratory toxicity to common avian test species (2) laboratory toxicity to common mammalian test species (3) species-specific field or laboratory toxicity data	avian NOEL mammal NOEL
No impacts to endangered or protected piscivorous wildlife species (e.g., migratory birds)	(c) No probability for any reduction in populations of protected piscivorous wildlife species	(1) laboratory toxicity to common avian test species (2) laboratory toxicity to common mammalian test species (3) species-specific field or laboratory toxicity data	avian NOEL mammal NOEL

[a] A 10% level of population effects is approximately the limit of detection of field measurement techniques and is likley below the detection limits of the public (e.g., catch-and-release fishermen).

4.0 EXPOSURE CHARACTERIZATION

Exposure assessment attempts to quantify the magnitude or type of actual and/or potential exposures of ecological receptors to site-specific stressors, in this case COCs. This part of the assessment includes quantification of COC release, transport and fate, ecological receptor characterization, and determination (either by measurement or modeling) of exposure point concentrations. This section is a brief explanation of the rationale and methods for quantification of contaminant levels, selection of significant ecological receptors, and determination of exposure point concentrations.

4.1 Transport and Fate Estimation

Data on current locations and concentrations of COCs were determined by direct sampling of abiotic media upstream of, immediately adjacent to, and downstream of the Facility. Sampling downgradient of site boundaries provided limited quantitative measurements of COC migration phenomena. Exposure point concentrations in sediments and surface water represent the upper 95th percentile of the geometric mean of measured concentrations in these media; i.e., a reasonable maximum exposure (RME). The intent of this approach is to estimate a conservative exposure case (i.e., well above the average case) that is still within a range of possible exposures.

4.2 Ecological Receptors

This screening assessment involves determining whether site-related COCs could cause potential adverse effects to these particular species. Since evaluating risks posed by COCs to each and every species or population present or potentially present is not feasible, an assessment must focus on a limited number of receptors. This subset of potential ecological receptors (termed "indicator species") may include organisms which are: (a) chronically exposed to site-related chemicals, (b) endangered, threatened, special concern or protected species, (c) of relevance to assessment endpoints, and (d) chronically exposed via a pathway which is different from previously considered organisms.

The following species were selected as indicator species: phytoplankton, aquatic macrophytes, zooplankton, pelagic invertebrates (aquatic insects), benthic invertebrates (oligochaetes), fish (white suckers), bullfrog, snapping turtle, mallard duck, raccoon, and great blue heron. They are interrelated by a site-specific food web as shown in Figure 4-1. These species were selected as indicators because: (a) they were observed in or near the Facility or study area, (b) they filled a niche not accounted for in the food web by other species, (c) suitable habitat was available for these species, even if they were not observed at the site during field surveys, (d) they represent either top predators, top predator prey species, or protected species, and/or (e) toxicity data was available for a number of COCs.

4.3 Estimated Receptor Exposures

For the purposes of this screening assessment, exposure estimates were calculated only for generic invertebrates, generic fish, and a representative piscivorous species (great blue heron [*Ardea herodias*]). Aquatic invertebrates and fish in river waters were assumed to be primarily and directly exposed to COCs by osmotic exchange with surrounding surface waters. Surface water concentrations are affected by variables such as dilution and it is a mobile media not necessarily directly related to the fixed sediments where the greatest mass of COCs is entrained. Thus pore water concentrations were taken as exposure

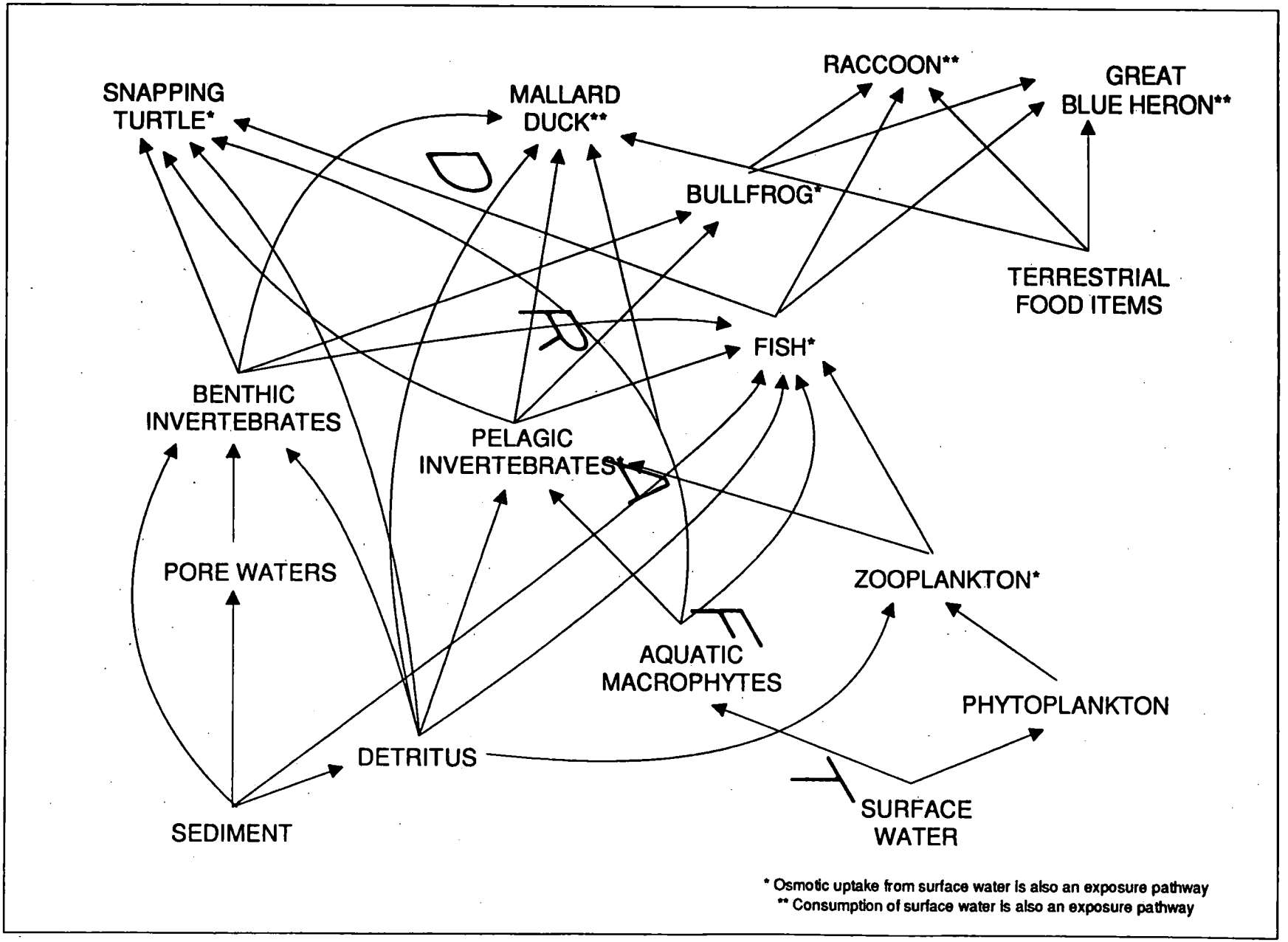


FIGURE 4-1: PAWTUXET RIVER FOOD WEB

point concentrations for aquatic biota, to provide an estimate of toxicity more closely related to sediments. Pore water concentrations were estimated as a function of sediment concentrations, assuming equilibrium conditions between sediment solids and sediment pore waters. Secondary, indirect exposure can occur through consumption of food items and incidental ingestion of contaminated particulates; however, for this assessment contributions from the food web were assumed to be negligible. A simple, conservative model was used to derive exposure point concentrations for pore water from sediments, in that:

$$EPC_{aq} = C_{pw} = C_{sed} / (K_{oc} \times f_{oc}) \qquad \text{[Eq. 4-1]}$$

where: EPC_{aq} = COC exposure point concentration for aquatic receptors (fish and invertebrates) (mg/L), C_{pw} = COC concentration in pore water (mg/L), C_{sed} = COC environmental concentration in sediment (mg/kg), K_{oc} = soil/water partition coefficient normalized for organic carbon (unitless), and f_{oc} = fractional organic matter content of the sediment. Measured sediment f_{oc} values were 0.011 for upstream stations, 0.085 for Facility reach stations, and 0.0037 for downstream stations. Equation [4-1] provides a highly conservative estimate of exposure in that it assumes benthic macroinvertebrate and fish populations are exposed to theoretical pore water concentrations. Such an assumption is appropriate only for a screening-level assessment. This model also assumes that fish are continuously exposed to C_{pw} concentrations within each reach for their entire lifespans; this is a highly conservative assumption given what is known about the lack of home range fidelity in the two dominant species, white suckers and carp.

Soil/water partition coefficients normalized for organic carbon (K_{oc}) for inorganics (metals) were calculated using the relationship:

$$K_{oc} = K_d / f_{oc} \qquad \text{[Eq. 4-2]}$$

where: K_d = soil sorption coefficient, obtained from Baes et al., 1984. Partition coefficient values for organic COCs were calculated as follows (USEPA, 1993):

$$\log(K_{oc}) = 0.00028 + 0.983 \times \log(K_{ow}) \qquad \text{[Eq. 4-3]}$$

where K_{ow} = COC-specific octanol-water partitioning coefficient (unitless). Pore-water concentrations of inorganic (metals) and neutral (non-ionic) organic chemicals were calculated using Equation 4-1 (after USEPA, 1993; OWRS, 1989). It is assumed that ionic organics behave like neutral organics, only partitioning between water and sediment organic matter. This is a conservative assumption because partitioning to other phases would lower the aqueous concentration.

Higher trophic level species in the aquatic food web (e.g., mallard ducks, great blue heron, raccoon), not necessarily in direct contact with contaminated media, are exposed primarily through consumption of contaminated prey. Direct consumption of, or contact with, contaminated surface water was assumed to be negligible. Doses received by the great blue heron through consumption of contaminated prey items were determined using the following simple models (Landrum et al., 1992):

$$EPC_h = [(C_{pw} \times BCF) \times R_h \times \alpha \times \%] / BW_h \quad \text{[Eq. 4-4]}$$

where: EPC_h = exposure point concentration (applied daily dose) for great blue heron from consumption of contaminated aquatic prey species (mg/kg-day), BCF = COC-specific bioconcentration factor (L/kg), R_h = great blue heron ingestion rate of food = $0.648 \times BW_h^{0.651}$ = 0.118 (kg/day) (Nagy, 1987), α = COC-specific assimilation efficiency (unitless), BW_h = median adult great blue heron body weight = 2.97 (kg) (Palmer, 1962), and % = fraction of aquatic species in diet = 1.0 for great blue heron. Organic compounds were assigned a default α value of 0.9 for organics, while values for inorganics (metals) were assigned as follows (after Owen, 1989): As = 0.98, Ag = 0.5, Cr = 0.01, Cu = 0.5, Hg = 0.15, Ni = 0.05, Pb = 0.1, V = 0.5, and Zn = 0.5.

Bioconcentration factor values for inorganic COCs were obtained from EPA (1986) and were calculated for organic COCs as follows (Lyman et al., 1982):

$$\log (BCF) = 0.76 \times \log (K_{ow}) - 0.23 \quad \text{[Eq. 4-4]}$$

where K_{ow} = COC-specific octanol-water partitioning coefficient (unitless).

Calculated exposure point concentration values for indicator COCs in fish, invertebrates, raccoon, and great blue heron are summarized in Table 4-1.

TABLE 4-1
Estimated Exposure Point Concentrations for Aquatic and Terrestrial Receptors

CONSTITUENT OF CONCERN	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS		
	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)
INORGANICS									
Arsenic	1.70E+01	8.52E-02	1.46E-01	3.61E+01	1.81E-01	3.09E-01	1.05E+01	5.25E-02	8.99E-02
Barium	8.24E+01	1.37E+00	DG	3.80E+02	6.33E+00	DG	8.05E+01	1.34E+00	DG
Chromium	4.96E+01	5.84E-02	3.71E-04	1.26E+03	1.48E+00	9.42E-03	4.16E+01	4.90E-02	3.11E-04
Copper	9.76E+01	2.79E+00	1.11E+01	1.08E+03	3.09E+01	1.23E+02	3.99E+01	1.14E+00	4.53E+00
Cyanide	ND	ND	ND	1.17E+01	DG	DG	ND	ND	ND
Lead	1.73E+02	1.92E-01	3.74E-02	8.29E+02	9.21E-01	1.79E-01	1.28E+02	1.42E-01	2.77E-02
Mercury	1.33E-01	1.33E-02	4.37E-01	2.80E+00	2.80E-01	9.18E+00	6.15E-02	6.15E-03	2.02E-01
Nickel	2.95E+01	1.97E-01	1.84E-02	1.48E+02	9.85E-01	9.20E-02	1.23E+01	8.19E-02	7.64E-02
Silver	ND	ND	ND	2.14E+00	4.76E-02	2.91E+00	ND	ND	ND
Tin	ND	ND	ND	1.58E+01	3.50E-01	6.96E-03	ND	ND	ND
Vanadium	1.45E+01	1.45E-02	8.07E-03	4.94E+01	4.94E-02	2.75E-02	7.00E+00	7.00E-03	3.90E-03
Zinc	2.29E+02	5.73E+00	5.35E+00	1.39E+04	3.48E+02	3.24E+02	1.66E+02	4.15E+00	3.87E+00
ORGANICS									
1,2-Dichlorobenzene	1.20E-01	4.96E-03	4.01E-02	1.01E+00	5.41E-03	4.37E-02	ND	ND	ND
1,4-Dichlorobenzene	ND	ND	ND	1.46E+00	8.01E-03	6.36E-02	ND	ND	ND
Chlorobenzene	5.11E-01	7.50E-02	2.27E-01	2.12E+00	4.03E-02	1.22E-01	8.11E-02	3.54E-02	1.07E-01
Tinuvin 328	ND	ND	ND	1.68E+01	DG	DG	1.20E+00	DG	DG
Toluene	9.35E-02	1.93E-02	4.49E-02	8.60E+02	2.29E+01	5.35E+01	8.87E-02	5.43E-02	1.27E-01
Xylene (m & p)	2.10E-02	1.96E-03	8.43E-03	4.93E-01	5.95E-03	2.56E-02	ND	ND	ND
Xylene (o)	7.00E-02	6.53E-03	2.81E-02	2.30E-01	2.78E-03	1.20E-02	ND	ND	ND
PESTICIDES									
2,4-D	ND	ND	ND	2.02E-01	4.10E-03	1.18E-02	ND	ND	ND
4,4'-DDE	ND	ND	ND	5.43E-02	1.45E-06	7.05E-04	6.28E-03	3.86E-06	1.87E-03
4,4-DDT	7.00E-03	8.42E-07	6.21E-04	8.15E-02	1.27E-06	9.37E-04	6.20E-03	2.22E-06	1.64E-03
BHC, alpha-	ND	ND	ND	3.79E-02	8.01E-05	1.33E-03	ND	ND	ND
Chlordane, gamma-	2.30E-02	8.57E-06	2.64E-03	5.09E-02	2.45E-06	7.55E-04	4.10E-03	4.54E-06	1.40E-03
Dieldrin	ND	ND	ND	4.51E-02	5.06E-05	1.37E-03	9.89E-03	2.55E-04	6.88E-03
Disulfoton	ND	ND	ND	1.20E-01	1.89E-04	3.93E-03	1.83E-02	6.61E-04	1.37E-02
Endrin	ND	ND	ND	1.00E-02	3.68E-07	1.40E-04	ND	ND	ND
Heptachlor	1.30E-02	1.44E-05	1.91E-03	4.40E-02	6.29E-06	8.36E-04	1.16E-02	3.82E-05	5.08E-03
Pentachlorophenol	ND	ND	ND	6.38E+00	8.92E-04	1.21E-01	ND	ND	ND
PAHs									
Acenaphthylene	ND	ND	ND	1.10E-01	1.81E-04	3.64E-03	ND	ND	ND
Anthracene	2.85E-01	1.22E-03	5.68E-02	9.30E-01	5.17E-04	2.40E-02	1.60E-01	2.05E-03	9.51E-02
Benzo(a)anthracene	1.70E+00	4.83E-04	1.83E-01	3.80E+00	1.40E-04	5.30E-02	7.20E-01	6.08E-04	2.31E-01
Benzo(a)pyrene	1.79E+00	1.02E-04	1.34E-01	4.10E+00	3.02E-05	3.97E-02	7.23E-01	1.22E-04	1.61E-01

TABLE 4-1
Estimated Exposure Point Concentrations for Aquatic and Terrestrial Receptors

CONSTITUENT OF CONCERN	UPSTREAM STATIONS			FACILITY STATIONS			DOWNSTREAM STATIONS		
	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)	Sediment EPC (mg/kg)	EPCaq Eq. 4-1 (mg/L)	EPCh Eq. 4-4 (mg/kg)
Benzo(b)fluoranthene	3.30E+00	1.04E-04	2.16E-01	8.90E+00	3.64E-05	7.55E-02	1.25E+00	1.17E-04	2.43E-01
Benzo(g,h,i)perylene	1.96E+00	1.39E-05	9.16E-02	4.80E+00	4.41E-06	2.90E-02	7.95E-01	1.68E-05	1.10E-01
Benzo(k)fluoranthene	3.70E+00	6.35E-05	2.11E-01	9.30E+00	2.07E-05	6.87E-02	1.43E+00	7.30E-05	2.43E-01
Chrysene	2.40E+00	6.82E-04	2.59E-01	6.90E+00	2.54E-04	9.63E-02	1.40E+00	1.18E-03	4.49E-01
Dibenzofuran	1.20E-01	9.72E-04	2.77E-02	3.20E-01	3.35E-04	9.55E-03	6.70E-02	1.61E-03	4.59E-02
Fluoranthene	5.82E+00	3.05E-03	7.22E-01	1.40E+01	9.49E-04	2.24E-01	3.60E+00	5.60E-03	1.33E+00
Fluorene	1.80E-01	1.27E-03	4.03E-02	7.90E-01	7.23E-04	2.29E-02	1.00E-01	2.10E-03	6.65E-02
Indeno(1,2,3-cd)pyrene	1.60E+00	4.29E-06	5.99E-02	4.30E+00	1.49E-06	2.08E-02	6.84E-01	5.45E-06	7.61E-02
Naphthalene	9.30E-02	4.21E-03	3.17E-02	2.95E+00	1.73E-02	1.30E-01	2.30E-01	3.09E-02	2.33E-01
Phenanthrene	1.60E+00	6.00E-03	3.10E-01	5.49E+00	2.67E-03	1.38E-01	8.62E-01	9.61E-03	4.96E-01
Pyrene	2.30E+00	1.69E-03	3.08E-01	7.72E+00	7.35E-04	1.34E-01	1.12E+00	2.46E-03	4.47E-01
PCBs									
PCB-1248	ND	ND	ND	5.96E-01	1.56E-05	7.70E-03	ND	ND	ND
PCB-1254	ND	ND	ND	1.49E+00	2.07E-05	1.67E-02	ND	ND	ND
PHTHALATE ESTERS									
Di-n-butylphthalate	3.30E-02	2.32E-05	4.37E-03	1.45E+00	1.32E-04	2.49E-02	5.30E-02	1.11E-04	2.09E-02
Di-n-octylphthalate	ND	ND	ND	1.41E+00	1.50E-08	3.10E-03	9.77E-01	2.39E-07	4.93E-02
DIOXINS/FURANS									
HxCDD	ND	ND	ND	4.10E-03	1.43E-08	3.36E-05	ND	ND	ND
HxCDF	ND	ND	ND	3.27E-03	1.14E-08	2.67E-05	ND	ND	ND
PeCDD	ND	ND	ND	3.62E-04	1.26E-09	2.96E-06	ND	ND	ND
PeCDF	ND	ND	ND	1.31E-03	4.57E-09	1.07E-05	ND	ND	ND
TCDD	ND	ND	ND	3.18E-04	1.11E-09	2.61E-06	ND	ND	ND
TCDF	ND	ND	ND	3.11E-04	1.09E-09	2.55E-06	7.67E-05	6.15E-09	1.44E-05

ND = not detected
DG = data gap; parameter required to calculate exposure value is not available
EPC = exposure point concentration; EPCaq = EPC for aquatic receptors; EPCh = EPC for piscivorous avifauna (heron)

5.0 ECOLOGICAL EFFECTS CHARACTERIZATION

The potential for adverse effects was addressed using four approaches: (a) analysis of benthic invertebrate community structure, (b) results of bioassay testing of sediment, surface water, and pore water, (c) comparison of observed exposure point concentrations to previously published effect levels for terrestrial and aquatic animals, and (d) other observed effects.

5.1 Effects Assessment

5.1.1 Ecotoxicological Analysis

A toxic COC may either kill an organism outright (acute effect) or provoke less obvious adverse damage such as reduced fecundity, reduced growth, damage to some organ, or low levels of mortality (chronic effects). A NOAEL is the dose or concentration at or below which no adverse effects have been observed in exposed animals and is one to which a population of organisms may be exposed with no adverse impacts on any individuals. A NOAEL is an acceptable level for this assessment.

Dose-response values were obtained from the literature for all indicator species-COC combinations appropriate to this assessment. In addition to hardcopy literature searches, the following commercial on-line electronic databases were also queried: AQUIRE, TOXNET (Toxicology Data Network), HSDB (Hazardous Substances Data Bank), RTECS (Registry of Toxic Effects of Chemical Substances), Zoological Record Online, TOXLINE, NTIS, BIOSIS Previews, Conference Papers, ENVIROLINE, Environmental Bibliography, Life Sciences Collection, and PHYTOTOX. Specific searches were carried out for all indicator species.

When data were available concerning toxicity of a COC to fish, avian, or mammalian indicator species, the highest NOEL derived from a chronic study using an indicator species or a taxonomically similar species was the preferred test endpoint and was used as the toxicity reference value (TRV). When literature data (particularly NOEL values) were not available for a given COC-indicator species combination, acceptable TRVs were extrapolated from other test endpoints (usually median lethal dose (LD50), median lethal concentration (LC50), effective concentration (EC50), lowest observed effect level (LOEL) or lowest observed adverse effect (LOAEL) values), and from toxicological studies on other, more common, test species. NOEL values were used directly as TRVs, while any acute effect level (LD50, LC50, EC50) was divided by an uncertainty factor of 100 to derive a TRV, and any LOEL or LOAEL was divided by an uncertainty factor of 10 to estimate a TRV (Sloof et al., 1986; Suter, 1993; Urban and Cook, 1986). Toxicity reference values derived for aquatic receptors are summarized in Table 5-1.

5.1.2 Laboratory Studies

Surface water and sediments from the Pawtuxet River near the Facility were bioassayed for toxicity and details of these tests are reported elsewhere (IT, 1991a,b). Figure 5-1 graphically illustrates mean survival along a station gradient using a subset of results from the sediment and pore water bioassay tests. In summary, surface waters did not produce significant mortality in water fleas (*Ceriodaphnia dubia*) or fathead minnows (*Pimephales promelas*) upstream, adjacent to, or downstream from the Facility. Reproduction among *C. dubia* was not affected by surface water at the Facility but was slightly decreased by water from locations immediately upstream and downstream from the Facility, compared with laboratory

TABLE 5-1
Toxicity Reference Values for Terrestrial and Aquatic

CONSTITUENT OF CONCERN	Test Species	Test Species Type[a]	Toxicity Endpoint[b]	Toxicity Reference Value[c]	Reference	Notes
INORGANICS						
Arsenic (III)	*Ratus ratus*	MA	oral LD-50	6.00E+00	USPHS, 1987	
	Lepomis macrochrius	FH	96hr LC-50	1.62E-01	LeBlanc, 1980	
	various	IV	NOEL	1.90E-01	Suter, et al., 1992	
Barium	*Ratus ratus*	MA	NOEL	5.00E+00	Schroeder and Mitchener, 1975	
	various	FH	NOEL	1.09E-01	Suter et al., 1992	NAWQC chonic criteria estimate
	various	IV	NOEL	1.09E-01	Suter et al., 1992	NAWQC chonic criteria estimate
Chromium (III)	*Anas sp.*	MA	LOAEL	1.00E+00	Eisler, 1986	
	Nuria denricus	FH	96hr LC-50	2.90E-01	Abbasi and Soni, 1984	
	Daphnia magna	IV	NOEL	2.10E-01	Suter et al., 1992	NAWQC chonic criteria estimate
Copper	non-ruminant animals	MA	NOEL	1.00E+02	NRC, 1980; Suter, 1991	
	Lepomis macrochirus	FH	96hr LC-50	3.20E-04	Thompson et al., 1980	
	Ephemerella subvaria	IV	96hr LC-50	3.20E-03	Warnick and Bell, 1969	
Cyanide	*Anas* sp.	MA	LD-50	1.43E-02	Eisler, 1991	
	various	FH	NOEL	2.20E-02	Suter et al., 1992	NAWQC chonic criteria
	various	IV	NOEL	5.20E-03	Suter et al., 1992	NAWQC chonic criteria
Lead	*Falco sparvenius*	MA	LOAEL	5.00E+00	Hoffman et al., 1985ab	
	Lepomis macrochirus	FH	96hr LC-50	2.38E-01	LeBlanc, 1980	
	Asellus aquaticus	IV	96hr LC-50	6.41E-01	Martin and Holdich, 1986	
Mercury	*Anas platyrhyncus*	MA	NOAEL	5.50E-01	Heinz, 1974	
	Carassius auratus	FH	LOAEL	3.00E-03	Weir and Hine, 1970	
	Asellus aquaticus	IV	96hr LC-50	1.99E-03	Martin and Holdich, 1986	
Nickel	*Ratus ratus*	MA	LD-50	3.50E+00	NRC, 1980	
	Carassius auratus	FH	LC-50	2.80E-02	Seiler et al., 1988	
	Asellus aquaticus	IV	96hr LC-50	1.19E+00	Martin and Holdich, 1986	
Silver	*Ratus ratus*	MA	NOEL	1.00E+02	NRC, 1980	
	Lepomis macrochirus	FH	96hr LC-50	1.30E-04	Holcombe et al., 1987	
	Daphnia magna	IV	48hr LC-50	1.10E-04	Mount and Norberg, 1984	
Tin	*Ratus ratus*	MA	NOEL	1.00E+00	Eisler, 1989	
	various	FH	LOEL	3.50E-02	Suter, 1991	
	Crangonyx pseudogracilis	IV	96hr LC-50	5.01E-01	Martin and Holdich, 1986	
Vanadium	*Ratus ratus*	MA	NOEL	5.00E+00	Schroeder and Mitchener, 1975	
	various	FH	NOEL	8.00E-02	Suter et al., 1992	
	Crangonyx pseudogracilis	IV	96hr LC-50	1.23E-01	Martin and Holdich, 1986	
Zinc	*Coturnix c. japonica*	MA	LC-50	9.90E+00	Eisler, 1993	
	Lepomis macrochirus	FH	NOEL	1.10E-01	Suter et al., 1992	NAWQC chonic criteria
	Asellus aquaticus	IV	NOEL	1.10E-01	Suter et al., 1992	NAWQC chonic criteria

TABLE 5-1
Toxicity Reference Values for Terrestrial and Aquatic

CONSTITUENT OF CONCERN	Test Species	Test Species Type[a]	Toxicity Endpoint[b]	Toxicity Reference Value[c]	Reference	Notes
ORGANICS						
1,2-Dichlorobenzene	*Lepus* sp.	MA	LD-50	5.00E+00	Sax, 1984	
	Lepomis macrochirus	FH	48hr LC-50	5.60E-02	Buccafusco et al., 1981	
	Daphnia magna	IV	EC-50	1.70E-02	Sheedy et al., 1991	
1,4-Dichlorobenzene	*Oryctolagus cuniculus*	MA	LD-50	9.60E-01	Verschueren, 1983	
	Oncorhynchus mykiss	FH	96hr LC-50	8.80E-03	Mayer and Ellersieck, 1986	
	Daphnia magna	IV	EC-50	3.20E-01	Sheedy et al., 1991	
Chlorobenzene	*Ratus ratus*	MA	LD-50	3.40E+01	Kimura et al., 1971	benzene value
	Lepomis macrochirus	FH	96hr LC-50	1.60E-01	Buccafusco et al., 1981	
	various	IV	NOEL	1.16E-02	Suter et al., 1992	NAWQC chonic criteria estimate
Tinuvin 328		MA		MD		
	Brachydanio rerio	FH	96hr LC-50	1.00E+00	CIBA data	
	Daphnia sp.	IV	24hr EC-50	1.00E+00	CIBA data	
Toluene	*Ratus ratus*	MA	NOAEL	8.00E+02	Ungvary et al., 1982	
	Lepomis macrochirus	FH	96hr LC-50	7.40E-01	Johnson and Finley, 1980	
	Daphnia magna	IV	NOEL	1.04E-02	Suter et al., 1992	
Xylene (total)	*Ratus ratus*	MA	LD-50	4.30E+01	Jori et al., 1986	
	Catostomus commersoni	FH	96hr LC-50	1.61E-01	Holcombe et al., 1987	
	Daphnia magna	IV	48hr LC-50	3.82E-02	Holcombe et al., 1987	
PESTICIDES						
2,4-D	*Tyto alba*	MA	NOAEL	5.00E-01	Mendenhall et al., 1983	value for dieldrin
	Lepomis macrochirus	FH	48hr LC-50	9.00E-03	Verschueren, 1983	
	Pteronarcys spp.	IV	96hr LC-50	7.00E-05	Johnson and Finley, 1980	DDT value
4,4'-DDE	*Anas platyrhyncus*	MA	LC-50	3.57E+01	Hill et al., 1975	
	Lepomis macrochirus	FH	96hr LC-50	2.40E-03	Stuart, 1975	
	Pteronarcys spp.	IV	96hr LC-50	7.00E-05	Johnson and Finley, 1980	DDT value
4,4'-DDT	*Pelecanus occidentalis*	MA	LOEL	1.50E-02	Anderson, 1975	
	Lepomis macrochirus	FH	96hr LC-50	1.60E-05	Johnson and Finley, 1980	
	Pteronarcys spp.	IV	96hr LC-50	7.00E-05	Johnson and Finley, 1980	
BHC, alpha-	*Anas platyrhyncus*	MA	LD-50	2.00E+01	Stuart, 1975	BHC, gamma- value
	Lepomis macrochirus	FH	96hr LC-50	2.50E-04	Johnson and Finley, 1980	BHC, gamma- value
	Pteronarcys californica	IV	96hr LC-50	4.50E-05	Johnson and Finley, 1980	BHC, gamma- value
Chlordane, gamma-	*Tyto alba*	MA	LD-50	7.50E-01	Eisler, 1990	chlordane value
	Lepomis macrochirus	FH	96hr LC-50	5.70E-04	Mayer and Ellersieck, 1986	chlordane value
	Pteronarcys californica	IV	96hr LC-50	1.50E-04	Mayer and Ellersieck, 1986	chlordane value
Dieldrin	*Tyto alba*	MA	NOAEL	5.00E-01	Mendenhall et al., 1983	
	Lepomis macrochirus	FH	96hr LC-50	3.10E-05	Johnson and Finley, 1980	

TABLE 5-1
Toxicity Reference Values for Terrestrial and Aquatic

CONSTITUENT OF CONCERN	Test Species	Test Species Type[a]	Toxicity Endpoint[b]	Toxicity Reference Value[c]	Reference	Notes
	Pteronarcys spp.	IV	96hr LC-50	5.00E-05	Johnson and Finley, 1980	
Disulfoton	*Tyto alba*	MA	NOAEL	5.00E-01	Mendenhall et al., 1983	value for dieldrin
	Lepomis macrochirus	FH	96hr LC-50	6.30E-04	Verschueren, 1983	
	Gammarus fasciatus	IV	96hr LC-50	2.10E-04	Verschueren, 1983	
Endrin	*Tyto alba*	MA	NOAEL	5.00E-01	Mendenhall et al., 1983	value for dieldrin
	Lepomis macrochirus	FH	96hr LC-50	2.30E-05	Thurston et. al., 1985	
	Daphnia magna	IV	48hr LC-50	8.80E-04	Thurston et. al., 1985	
Heptachlor	*Ratus ratus*	MA	LD-50	4.00E-01	Sax, 1992	
	Lepomis macrochirus	FH	96hr LC-50	5.30E-05	Johnson and Finley, 1980	
	Pteronarcys californica	IV	96hr LC-50	1.10E-05	Johnson and Finley, 1980	
Pentachlorophenol	*Coturnix c. japonica*	MA	LC-50	5.20E+01	Hill et al., 1975	
	Lepomis macrochirus	FH	96hr LC-50	3.20E-04	Johnson and Finley, 1980	
	Daphnia magna	IV	48hr LC-50	1.45E-03	Berglind and Dave, 1984	
PAHs						
Acenaphthylene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH	NOEL	4.13E-01	Suter et al., 1992	lowest chronic value
		IV	NOEL	6.60E+00	Suter et al., 1992	lowest chronic value
Anthracene	*Agelaius phoeniceus*	MA	LD-50	1.11E+00	Schafer et al., 1983	
	various	FH	NOEL	9.00E-05	Suter et al., 1992	lowest chronic value
	various	IV	NOEL	2.10E-03	Suter et al., 1992	lowest chronic value
Benzo(a)anthracene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH	NOEL	2.70E-06	Suter et al., 1992	NAWQC chonic criteria estimate
	Daphnia magna	IV	NOEL	6.50E-04	Suter et al., 1992	lowest chronic value
Benzo(a)pyrene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Lepomis macrochirus	FH	NOEL	1.30E-06	Suter et al., 1992	NAWQC chonic criteria estimate
	Daphnia magna	IV	NOEL	3.00E-04	Suter et al., 1992	lowest chronic value
Benzo(b)fluoranthene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH		MD		
		IV		MD		
Benzo(g,h,i)perylene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH		MD		
	Daphnia magna	IV	LOEL	2.00E-05	Pilli et al., 1988	
Benzo(k)fluoranthene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH		MD		
	Daphnia magna	IV	LOEL	1.40E-04	Pilli et al., 1988	
Chrysene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
		FH		MD		

TABLE 5-1
Toxicity Reference Values for Terrestrial and Aquatic

CONSTITUENT OF CONCERN	Test Species	Test Species Type[a]	Toxicity Endpoint[b]	Toxicity Reference Value[c]	Reference	Notes
	Daphnia magna	IV	LOEL	7.00E-05	Pilli et al., 1988	
Dibenzofuran	*Agelaius phoeniceus*	MA	LD-50	1.02E+00	Schafer et al., 1983	
		FH	NOEL	2.00E-03	Suter et al., 1992	NAWQC chonic criteria estimate
		IV	NOEL	2.00E-03	Suter et al., 1992	NAWQC chonic criteria estimate
Fluoranthene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Lepomis macrochirus	FH	48hr LC-50	4.00E-01	Buccafusco et al., 1981	
	Neanthes arenaceodentata	IV	96hr LC-50	5.00E-03	Rossi and Neff, 1978	
Fluorene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Lepomis macrochirus	FH	96hr LC-50	9.10E-03	Finger et al., 1985	
	Daphnia magna	IV	96hr LC-50	4.30E-03	Mayer and Ellersieck, 1986	
Indeno(1,2,3-cd)pyrene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Lepomis macrochirus	FH	LOEL	5.00E-01	Pilli et al., 1988	value for benzo(a)pyrene
		IV		MD		
Naphthalene	*Mus musculus*	MA	LD-50	3.53E+00	Plasterer et al., 1985	
	Cyprinodon variegatus	FH	24hr LC-50	2.40E-02	Anderson et al., 1974	
	Daphnia magna	IV	96hr LC-50	8.60E-02	LeBlanc, 1980	
Phenanthrene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Gambusia affinis	FH	96hr LC-50	1.50E+00	EPA, 1970	
	Daphnia pulex	IV	NOEL	2.00E-01	Suter et al., 1992	lowest chronic value
Pyrene	*Ratus ratus*	MA	LD-50	5.00E-01	Suter, 1991	lowest PAH value
	Gambusia affinis	FH	96hr TLm	2.60E-05	Verschueren, 1983	
		IV		MD		
PCBs						
PCB-1248	*Mirounga angustirostris*	MA	LOEL	6.40E-02	Ringer, 1983; Suter, 1991	
	Lepomis macrochirus	FH	96hr LC-50	6.90E-03	Johnson and Finley, 1980	
	Daphnia magna	IV	96hr LC-50	2.60E-05	Nebecker and Puglisi, 1974	
PCB-1254	*Anas platyrhyncus*	MA	LC-50	2.70E+01	Hill et al., 1975	
	Lepomis macrochirus	FH	96hr LC-50	2.74E-02	Johnson and Finley, 1980	
	Ischnura spp.	IV	96hr LC-50	2.00E-03	Johnson and Finley, 1980	
PHTHALATE ESTERS						
Di-n-butylphthalate	*Ratus ratus*	MA	NOAEL	4.00E+03	Smith, 1982	
	Lepomis macrochirus	FH	96hr LC-50	7.30E-03	Buccafusco, et al., 1981	
	Daphnia magna	IV	LOEL	7.17E-01	Suter et al., 1992	
Di-n-octylphthalate	*Mus musculus*	MA	LD-50	6.51E+01	Antonuk, 1973	
	Pimephales promelas	FH	96hr LC-50	4.50E-04	DeFoe et al., 1990	ortho- isomer value
		IV		MD		
DIOXINS/FURANS						

TABLE 5-1
Toxicity Reference Values for Terrestrial and Aquatic

CONSTITUENT OF CONCERN	Test Species	Test Species Type[a]	Toxicity Endpoint[b]	Toxicity Reference Value[c]	Reference	Notes
TCDD	*Ratus ratus*	MA	LD-50	2.20E-04	Dickson and Buzik, 1993	used to represent toxicity
	Pimephales promelas	FH	28d LC-50	1.70E-08	Adams et al., 1986	of all dioxin/furan isomers
	Daphnia magna	IV	NOEL	1.33E-06	Isensee and Jones, 1975	and congeners

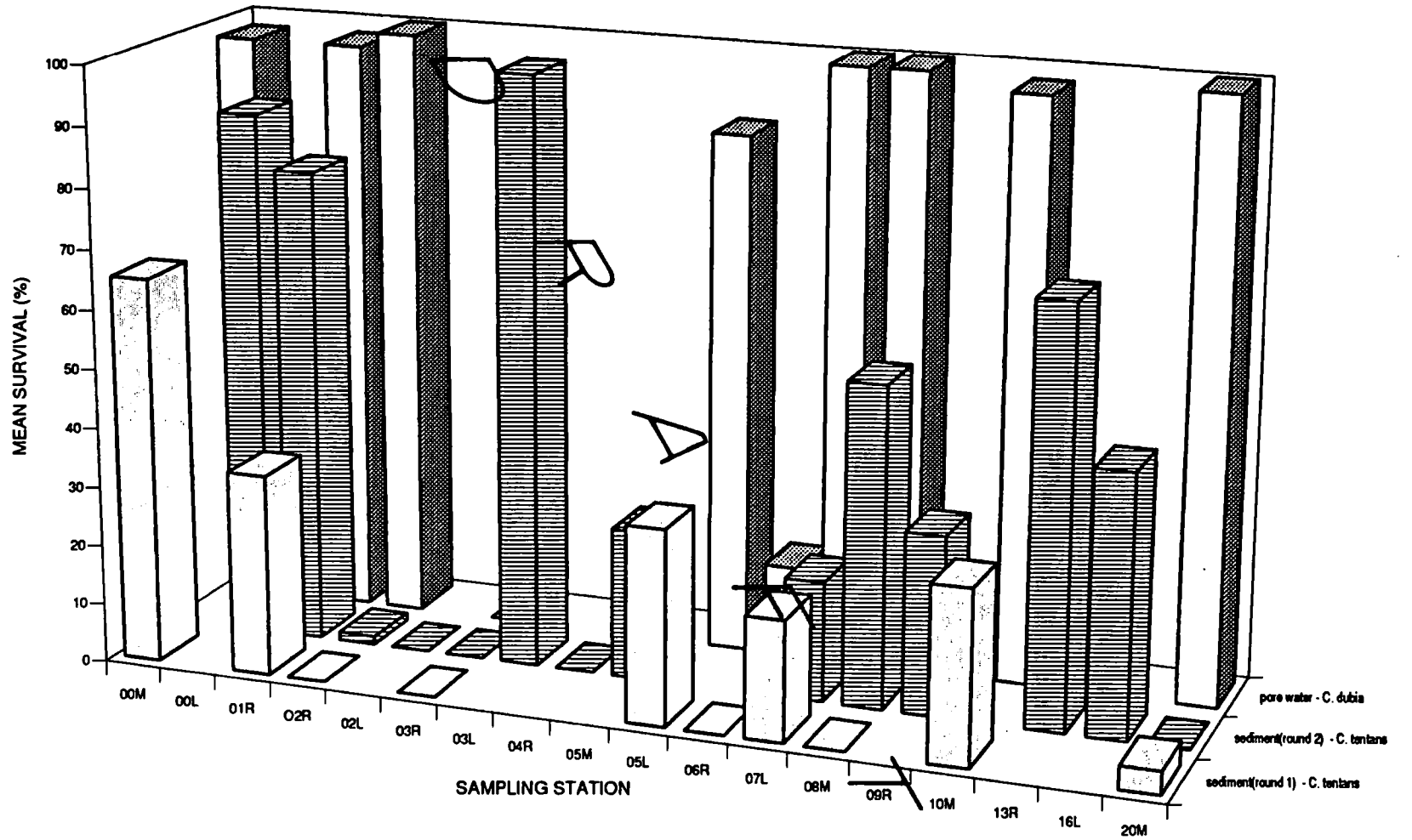
NOTES

[a] TEST SPECIES TYPES
MA = mammal or bird
FH = fish
IV = invertebrate

[b] ENDPOINT TYPES:
NOAEL = no observed adverse effect level
NOEL = no observed effect level
LD50 = dose that is lethal to 50% of test organisms
LC50 = concentration that is lethal to 50 % of test organisms
LOAEL = lowest observed adverse effect level
LOEL = lowest observed effect level
EC50 - adverse effect in 50% of test organisms
MD = missing data; no suitable toxicity data available to determine a TRV

[c] Toxicity Reference Value = NOEL = LD50/100 or LC50/100 or LOEL/10 or LOAEL/10 (mg/kg for MA test species; mg/L for FH, IV, or AM test species)

Figure 5-1
Summary of Sediment and Pore Water Bioassay Results



water. Pore water from sediments produced mortality in *C. dubia* at two of six locations sampled as the river flows through the Facility, while upstream and downstream pore water was not toxic. The generally elevated survivorship from exposure to pore water suggests that many toxic COCs are sediment-bound and not bioavailable. Sediments from a majority of stations adjacent to the Facility produced 100% mortality among exposed midges (*Chironomus tentans*), while sediment collected just upstream and downstream was also toxic but to a lesser extent. Sediment collected farther downstream produced significant toxicity and may indicate that another contaminant source is present in this reach of the river.

5.1.3 Field Investigations

Benthic macroinvertebrates inhabiting riverine sediments include insects, annelids, mollusks, flatworms, and crustaceans that may be omnivores, carnivores, or herbivores. In a well-balanced system, all three types are likely to be present. Trophic levels include deposit and detritus feeders, parasites, scavengers, grazers, and predators. These organisms are important members of aquatic food webs and their health is reflected in the health of higher aquatic vertebrates such as fish. The macroinvertebrate community in an aquatic ecosystem is very sensitive to stress and can serve as a useful tool for detecting either anthropogenic or natural environmental perturbations. Stress is generally reflected by a decrease in species richness (number of species/taxa present in a sample), a decrease in species diversity, and increased dominance of a few stress-tolerant species.

Samples of the benthos were collected with a Ponar grab sampler at stations upstream, adjacent to, and downstream of the Facility. Samples were enumerated and identified to the lowest practical taxon and resulting data analyzed using the following metrics: (a) species/taxa richness (S), which generally decreases with decreasing water quality, habitat diversity, and habitat suitability, (b) the Berger-Parker dominance index (D), which generally increases as a few stress-tolerant species begin to dominate and diversity in the macrobenthos diminishes (Berger and Parker, 1970; Magurran, 1988), and (c) the Shannon-Weiner diversity index (H') which tends to decrease as species are removed by pollution stress (Magurran, 1988).

Richness, dominance, and diversity index values are shown, by station, in Table 5-2; Figure 5-2 shows species richness along this station gradient in relation to the facility; Figure 5-3 shows dominance and diversity along this same gradient. Almost all assemblages sampled were dominated by tubifex worms, a pollution-tolerate species. Dominance is elevated and diversity depressed at Station B-02R within the Facility reach and, with the exception of Station B-02R, diversity appears to be higher within the Facility reach than downstream. The increase in dominance and the wide fluctuations in diversity below Station B-07L suggest, as did the bioassay results (Section 5.1.2), that another contaminant source may be impacting this reach of the river. A resemblance (community similarity) analysis (Table 5-3), based on chord distances between sampling stations, does not suggest the existence of any striking dissimilarities in intrastation faunal assemblages. No clear pattern of environmental stress in relation to the Facility is evident; however, these limited community analysis data strongly suggest that benthic macroinvertebrate communities in the Pawtuxet River are under some degree of stress throughout the length investigated.

White suckers collected in the Pawtuxet River ranged in length from 63 to 458 mm, while common carp

Table 5-2
Analysis of Spring Benthic Invertebrate Sampling Data

SAMPLING STATION	RICHNESS (S)	DOMINANCE (D)	DIVERSITY (H')
B-00-M	12	0.92	0.37
B-00-L	7	0.84	0.65
B-01-R	13	0.59	1.38
B-02-L	12	0.61	1.22
B-02-R	7	0.83	0.65
B-03-R	11	0.50	1.21
B-04-R	13	0.70	1.12
B-05-L	8	0.49	1.34
B-05-M	10	0.75	0.91
B-06-R	8	0.78	0.82
B-07-L	11	0.85	0.62
B-08-M	8	0.49	1.11
B-09-R	9	0.98	0.12
B-10-M	11	0.54	1.16
B-13-R	6	0.96	0.20
B-16-L	7	0.93	0.33
B-20-M	8	0.93	0.32

FIGURE 5-2
Benthic Speceies/Taxa Richness by Sampling Station (Smoothed Data)

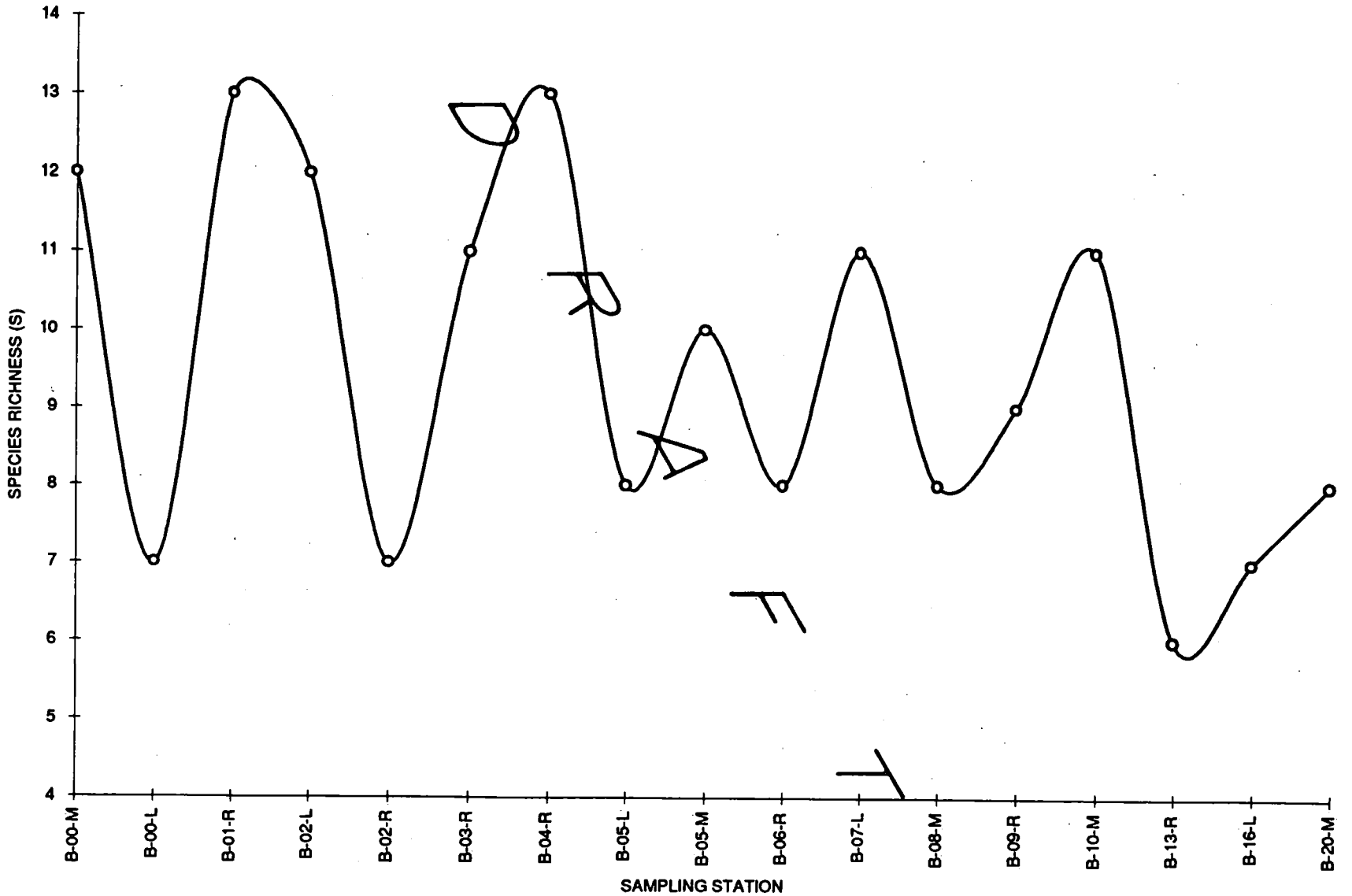


FIGURE 5-3
Species Dominance and Species Diversity by Sampling Station (Smoothed Data)

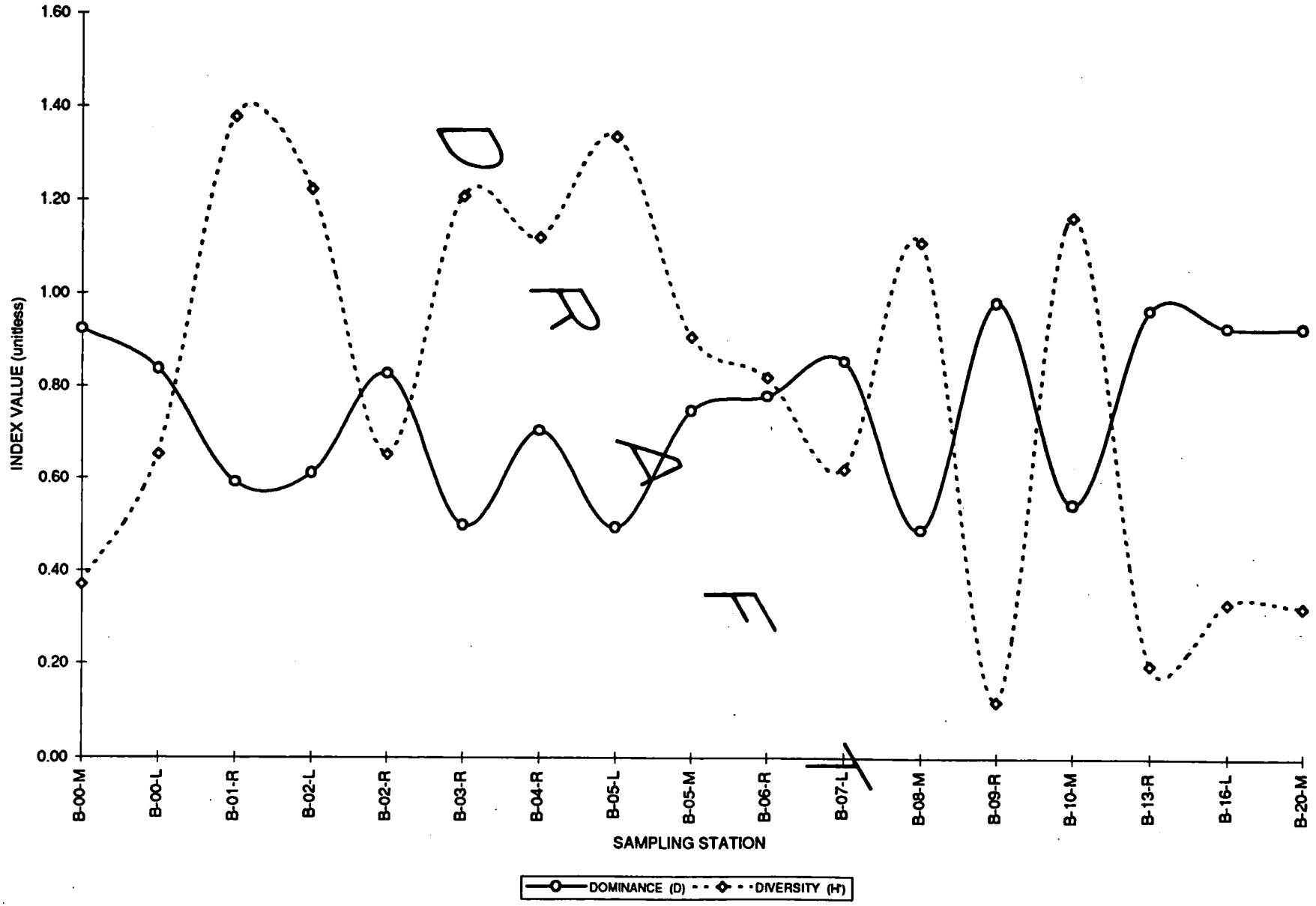


TABLE 5-3
CHORD DISTANCES BETWEEN PAWTUXET RIVER BENTHIC SAMPLING STATIONS

UPSTREAM STATIONS			FACILITY STATIONS									DOWNSTREAM STATIONS				
	00L	01R	02R	02L	03R	04R	05M	05L	06R	07L	08M	09R	10M	13R	16L	20M
00M	0.11	0.32	0.12	0.34	0.58	0.20	0.16	0.50	0.15	0.08	0.66	0.05	0.48	0.04	0.07	0.06
00L	0.25		0.03	0.26	0.50	0.18	0.10	0.48	0.07	0.04	0.60	0.10	0.39	0.10	0.12	0.08
01R			0.23	0.25	0.46	0.29	0.23	0.51	0.23	0.27	0.60	0.31	0.28	0.32	0.32	0.30
			02R	0.25	0.49	0.18	0.09	0.48	0.07	0.06	0.60	0.12	0.37	0.12	0.13	0.10
				02L	0.30	0.22	0.21	0.40	0.20	0.27	0.45	0.35	0.23	0.34	0.35	0.31
					03R	0.50	0.44	0.36	0.45	0.50	0.19	0.59	0.30	0.58	0.59	0.54
						04R	0.19	0.49	0.13	0.18	0.60	0.20	0.41	0.20	0.21	0.19
							05M	0.40	0.08	0.09	0.54	0.18	0.34	0.17	0.18	0.14
								05L	0.45	0.46	0.37	0.53	0.48	0.52	0.53	0.48
									06R	0.08	0.57	0.15	0.35	0.14	0.16	0.12
										07L	0.60	0.09	0.41	0.08	0.11	0.05
											08M	0.68	0.47	0.67	0.68	0.62
												09R	0.48	0.02	0.06	0.06
													10M	0.48	0.49	0.46
														13R	0.04	0.05
															16L	0.08

value for maximum dissimilarity between two stations is 1.41

ranged in length from 380 to 655 mm. White suckers were found to represent various age groups whereas the common carp were estimated to be about 10 to 15 years of age. The absence of carp cohorts ranging from first-of-year to about 10 years of age may indicate a lack of recruitment to the population, a lowering of fecundity for this species, or niche preemption for juveniles. Conversely, white suckers that were collected represent various cohorts and this population appears to be currently breeding in the Pawtuxet River.

Abnormalities noted on fish throughout the area investigated included external lesions, deformed fins, fin rot, black spot, leech parasitism, and scale pattern abnormalities. Approximately 51 percent of the common carp and 6 percent of the white suckers exhibited abnormalities; the cause for which is currently unknown. The lower frequency of abnormalities in white sucker, which were all younger than the carp, suggests that abnormalities noted in the common carp may have developed over some period of time.

5.2 Potential Ecological Effects

This is a discussion of adverse effects at the individual organism level due to exposure to various indicator COCs that may ultimately be manifested at higher levels of ecological organization (population, community). Where data were not available for every individual COC, effects were extrapolated from chemically similar COCs.

5.2.1 Inorganics

Chromium: Chromium is an essential trace element in humans and at least some laboratory animals; data are lacking for wild populations. Adverse effects have been documented for laboratory animals at 5.1 and 10.0 mg of Cr^{+6} and Cr^{+3}, respectively, per kilogram of diet (Eisler, 1986). High concentrations of Cr are normally found in RNA, but its role is unknown. Trace quantities are essential for carbohydrate metabolism in mammals as well as insulin action. In humans, a diet lacking Cr can lead to Cr deficiency. Half-life for elimination of Cr from rats is 0.5, 5.9 and 83.4 days (Mertz, 1969).

Hexavalent Cr (Cr^{+6}),was associated with adverse effects in invertebrates of widely separated taxa: reduced survival and fecundity of the cladoceran *Daphnia magna* at a concentration of 10 ppb and exposure for 32 days (USEPA, 1980b); growth inhibition of the protozoan *Chilomonas paramecium* at 1,100-3,000 ppb during exposures of 19-163 hours (Honig et al., 1980); abnormal movement patterns of larvae of the midge *Chironomus tentans* at 100 ppb in 48 hours (Catalan, 1982); and a temporary decrease in hemolymph glucose levels in the freshwater prawn *Macrobrachium lamarrei* surviving 1,840 ppb Cr^{+6} for 96 hours (Murti et al., 1983; Eisler, 1986).

Long-term exposure of rainbow trout for 180 days to high, but environmentally realistic, concentrations of 0.2 ppm Cr^{+6} resulted in elevated levels of Cr in kidney (3.5 mg/kg/fresh weight), liver (2.0 mg/kg), and muscle (0.6 mg/kg); after 90 days in Cr-free media, Cr levels were 1.6, 1.3, and 0.5, respectively (Calamari et al., 1982; Eisler, 1986). Sublethal effects were observed in freshwater teleosts following exposure to Cr^{+6}. In the snakehead fish (*Channa punctatus*), enzyme activities were altered in a wide variety of organs and tissues after exposure for 30 days to 2.6 ppm (Sastry and Sunita, 1984); the effects became life threatening after exposure for 120 days (Sastry and Tyagi; 1982, Sastry and Sunita 1982, 1983). Adverse effects of Cr to sensitive species have been documented at 10.0 µg/l (ppb) of Cr^{+6} and 30.0 µg/l of Cr^{+3} in

freshwater (Eisler, 1986).

Copper: This metal is widely distributed in water and is a naturally occurring element. Copper is an essential trace element to plants and animals (Callahan *et al.*, 1979), but becomes toxic at concentrations only slightly higher than essential levels (USEPA, 1985). According to Callahan *et al.* (1979), copper can be accumulated by biota but does not appear to be biomagnified. Bioaccumulation and biotransformation play an important role in the fate of copper. Copper toxicity to organisms is affected by several factors such as diet, age, loading density, and the chemistry of the water in which they live. Copper is relatively nontoxic to mammals, and tolerance limits are generally 10- to 100-fold higher than for aquatic fauna. Rabbits, ponies, and pigs can tolerate high levels, 300 to 800 mg/kg dry weight feed in their diets, with no toxicosis (Flemming and Trevors, 1989).

Copper occurs in natural waters primarily as the divalent cupric ion in free and complexed forms (Callahan et al., 1979). The cupric is highly reactive and forms moderate to strong complexes and precipitates with many inorganic and organic constituents of natural waters, like carbonates and phosphates. Free cupric ions are more toxic than most organic and inorganic Cu complexes, which tend to reduce toxicity attributable to total Cu (Andrew, 1976; Borgmann and Ralph, 1983). With this in mind, the interpretation of available toxicity data becomes complicated, because the proportion of free cupric ion present is highly variable and is difficult to measure except under carefully controlled laboratory conditions. Usually, data on Cu toxicity are reported using measurements other than total or dissolved Cu.

Copper is toxic to aquatic life at concentrations only slightly higher than those for plants and animals. Copper is known to act at cell surfaces to exert a toxic effect (MacLeod et al., 1967; Lamb and Tollefson, 1973). Most of the available tests on the toxicity of Cu to freshwater animals have been conducted with four salmonid (trout) species, fathead and bluntnose minnows, and bluegills. Acute values range from 6.5 µg/l for *Daphnia magna* in hard water to 10,200 µg/l for the bluegill in hard water (Cairns et al., 1978). Several factors are key contributors to the level at which Cu becomes toxic. These factors include water hardness, pH, and total organic carbon (TOC) level. As a general rule, Cu toxicity decreases with increases in alkalinity and TOC.

Chronic toxicity values are available for fifteen freshwater species. Values range from 3.873 µg/l for brook trout to 60.36 µg/l for northern pike *(Esox lucius)*. Fish and invertebrate species seem to be about equally sensitive to the chronic toxicity of Cu (USEPA, 1985).

Exposure of fresh water clam (*Corbicula fluminea*) to sublethal copper levels (0.012-0.25 mg/l) resulted in increased intracellular vacuolization of digestive diverticula, hemolytic infiltration, and increased mucocyte production in gills. Effects of starvation appeared at concentrations greater than 0.25 mg/l (Martin and Sparks, 1971). Imlay (1971) reported effects of copper on growth and reproduction of freshwater mussels at <0.025 ppm.

Lead: There are significant differences between species in response to lead poisoning and the effects are more pronounced with organic than with inorganic lead. Also, younger developmental stages are the most

sensitive and the effects are more severe at high temperatures and in diets deficient in minerals, fats, and proteins. Most of the information on the effects of lead to terrestrial invertebrates is concerned with the poisoning of waterfowl by lead shot (Clemens et al., 1975; cited in Eisler, 1988). Apparent symptoms of lead poisoning include loss of appetite and mobility, avoidance of other birds, lethargy, weakness, emaciation, tremors, dropped wings, green feces, impaired locomotion, loss of balance and depth perception, nervous system damage, inhibition of heme synthesis, damage to kidneys and liver, and death (Eisler, 1988; Mudge, 1983). Anemia, kidney disease, testicular and liver lesions, and neurological disorders have been associated with high brain lead concentrations in mourning doves (*Zeneida macroura*) (Kendall and Scanlon, 1982). Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively more tolerant to moderate lead exposure including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2000 ppm (Hoffman and Albers, 1984).

Lead adversely affects survival, growth, reproduction, development, and metabolism of most species under controlled conditions, but its effects are substantially modified by numerous physical, chemical, and biological variables (Eisler, 1988). In aquatic environments, dissolved Pb was the most toxic form. Effects of Pb toxicity on aquatic organisms were pronounced at elevated water temperatures, reduced pH, in younger life stages, after long exposures, and when organic Pb compounds were present (Eisler, 1988).

Adverse effects were noted on *Daphnia magna* reproduction at 1.0 µg Pb^{+2}/l. The exposure duration was 19 days and the reproductive impairment affected 10 percent of the study population (Eisler, 1988). At concentrations of 10µg Pb/l, 50 percent of the study population of *D. magna* showed reproductive impairment. Rainbow trout survival diminished at 3.5 µg of tetraethyllead per liter. The exposure duration for this experiment was 72 hours. An LC50 was reached at the above concentration (Eisler, 1988). Fathead minnows were not as sensitive to Pb as rainbow trout. An LC50 was reached in 96 hours at a concentration of 6,500 µg Pb^{+2}/l (Eisler, 1988).

Although Pb is concentrated by biota from water, there is no convincing evidence that it is transferred through food chains (Wong et al., 1978; USEPA, 1979; Branica and Konrad, 1980; Settle and Patterson, 1980; all cited in Eisler, 1988). In fact, Pb concentrations tended to decrease markedly with increasing trophic level in both detritus-based and grazing aquatic food chains (Wong et al., 1978; cited in Eisler, 1988). In the freshwater food chain of an alga *(Selenastrum capricornutum)*, to a daphnid *(Daphnia magna)*, to the guppy *(Poecilia reticulata)*, Pb accumulation progressively decreased from the alga to the guppy (Vighi, 1981; cited in Eisler, 1988).

Silver: Silver does not occur regularly in animal tissues. The major effect of excessive absorption of Ag is local or generalized impregnation of the tissues, where it remains as Ag sulfide. This forms an insoluble complex in elastic fibers, resulting in argyria (Goyer, 1986). Although the data for the systemic distribution of stable Ag are variable, they do not suggest that any organ or tissue, except perhaps the spleen, concentrates the element to any great extent (Coughtrey and Thorne, 1983). The National Research Council (NRC, 1980) set the maximum tolerable level for silver in animal food at 100 mg/kg based on studies of rats, chickens, and turkeys.

Silver exhibits oxidation states of 0, +1, +2, and +3, but only the 0 and +1 states occur to any extent in the environment. In natural water, the monovalent species is the form of environmental concern. Monovalent Ag ions may exist in various degrees of association with a large number of inorganic ions, such as sulfate, bicarbonate, and nitrate, to form numerous compounds with a range of solubilities and potentials for hydrolysis or other reactions (USEPA, 1980a). Most of the toxicity studies have been conducted with Ag nitrate, which is an excellent source of free soluble Ag ions.

Data concerning acute toxicity of Ag to freshwater organisms include 82 values for 10 species from nine different taxonomic families (USEPA, 1980a). Water hardness and chloride concentration are the two factors involved with acute Ag toxicity in aquatic organisms. For invertebrate species, acute values for Ag range from 0.25 µg/l for the water flea *Daphnia magna* to 4,500 µg/l for the scud *Gammarus pseudolimnaeus* (USEPA, 1980a). Acute values for fish range from 3.9 µg/l for the fathead minnow in soft water to 280 µg/l for rainbow trout in hard water. It appears that Ag is more toxic in soft water.

The available data indicate that acute toxicity to freshwater aquatic life may occur at concentrations of 1.2 µg/l in solution (water hardness of 50), and chronic toxicity at concentrations as low as 0.12 µg/l (USEPA, 1980a). Chronic values as high as 29 µg/l were determined in the laboratory. No information was found concerning the relationship between water hardness and chronic Ag toxicity.

Silver seems to bioaccumulate to some degree in food chains. The bioconcentration factors for Ag range from less than one for bluegills to 240 for insect larvae (USEPA, 1980a). Little information on bioaccumulation of Ag in food web matrixes exists. Limited information is available concerning the relationship of various forms of Ag and toxicity to aquatic animals.

Zinc: Zinc is readily transported in natural waters and is one of the most mobile of heavy metals (USEPA, 1987). Zinc dissolves faster in acidic waters and its toxicity to aquatic organisms is affected by pH and hardness. Zinc is an essential trace element in animal nutrition and can therefore be bioaccumulated by all organisms (Callahan et al., 1979). According to Phillips and Russo (1978), zinc becomes chronically toxic at levels close to those at which it begins to accumulate. Luten *et al.* (1987) found that zinc elimination in exposed bivalves apparently does not occur. Freshwater clams (*Anodonta californensis*) exposed to zinc showed continuous accumulation throughout a 36-day experiment (Pauley and Nakatani, 1968). Cladocerans are some of the most sensitive aquatic organisms to zinc (USEPA, 1987); the mean acute value was 0.094 ppm (hardness 50 ppm). Mean acute value for *Daphnia* sp. was 0.3 ppm and 89 ppm for *Argia* sp. (USEPA, 1987).

Beyer et al. (1985) found that very little of the Zn in soil was incorporated in flora and fauna; contamination came predominantly from aerial deposition. They also found higher concentrations of Zn in shrews and lower concentrations in mice, in contrast to Roberts and Johnson (1978), who found similar values between these insectivores and herbivores. Kidney concentrations in gray squirrels were higher in urban areas (25.5 to 31.9 µg/g) than in rural areas (14.3 to 18.6 µg/g) (McKinnin et al., 1976).

Zinc absorption is affected by numerous dietary factors. These interactions, and the uptake mechanisms,

are generally not well understood. In a laboratory study, Zn was administered in drinking water (200 mg/l) by itself and in combination with other metals (Cooke et al., 1990). Resultant Zn concentrations in the kidneys were higher than liver and femur concentrations. However, this was also the case when the combinations zinc/cadmium and iron/lead/zinc/cadmium were administered. In fact, the highest kidney concentrations occurred in the high Cd-only treatments. This may reflect the induction of metalliothioneins, which can bind Zn and Cd, and subsequent redistribution and accumulation in the kidney (Cooke et al., 1990).

Zinc seems to have a very low level of transfer potential through terrestrial food chains, which may be associated with its essential role in biological systems (Roberts and Johnson, 1978).

5.2.2 Organics

Chlorobenzene: Because the oral toxicity of these chemicals is poorly characterized, the relatively well studied compound hexachlorobenzene (HCB) is used as a model compound for this group. The acute lethal dose is 1000 mg/kg or greater (USEPA, 1980e; USEPA, 1984; NIOSH 1988). HCB causes liver damage in Japanese quail at 5 mg/kg in diet (USEPA, 1980e) and causes immunosuppression in mink and ferrets (by different criteria) at 25 mg/kg and 1 mg/kg in diet (Bleavins et al., 1983).

Toluene: Although toluene has not been shown to be carcinogenic or mutagenic in animals or humans, acute exposure to high levels of toluene can cause sublethal effects, particularly embyotoxic and fetotoxic effects (Clement Associates, 1985). Oral administration of toluene at doses of 260 mg/kg produced a significant increase in embryonic death in mice. The oral LD50 value for toluene in rodents is 2,000 mg/kg.

Aquatic organisms are relatively insensitive to toluene. EC50 and LC50 values for five freshwater species ranged from 12,700 to 313,000 ug/l (Clements Associates, 1985).

Xylene: Although no carcinogenic, mutagenic, or teratogenic effects of xylene have been identified in rats and mice, xylene has been shown to be fetotoxic in both species. Acute exposure to high levels of xylene can also cause sublethal effects, including central nervous system damage and irritation of mucous membranes in adult rats and mice (Clement Associates, 1985). The oral LD50 value for xylene in rodents is 2,000 mg/kg, and the LC50 for inhalation exposure is 13,000 mg/m^3.

Information on the toxicity of xylene to terrestrial wildlife or domestic animals is extremely limited. Because of the generally low acute toxicity of xylene observed in laboratory animals, it is unlikely that xylene would be highly toxic to wild or domestic birds and mammals (Clement Associates, 1985). However, quail eggs exposed to an aqueous solution of xylene applied to the egg surface showed decreased hatch rates and embryo viability at concentrations greater than 0.05 percent.

Some studies suggest that xylene adversely affects growth and survival of aquatic species. Xylene adversely affected adult trout at concentrations as low as 3.6 mg/l in a continuous flow system, and trout fry avoided xylene at concentrations greater than 0.1 mg/l (Clement Associates, 1985). The LD50 value for adult trout was determined to be 13.5 mg/l (Clement Associates, 1985). No Ambient Water Quality Criteria

have been established for acute or chronic freshwater exposure to xylene.

5.2.3 Pesticides

Chlorinated pesticides are persistent in the environment and volatilization, sorption to soil and sediments, and bioaccumulation are dominant fate processes (Callahan et al.,1979; Armstrong and Sloan, 1980). These highly lipophilic compounds are susceptible to large-scale transport due to their volatility. Plants absorb pesticides rapidly and efficiently through their leaves (Suns et al.,1981) and vertebrate uptake is affected by species, lipid levels, age, size, metabolic rate, reproduction, and feeding conditions (Moore and Ramamoorthy, 1984). Chlorinated pesticides provide control for target organisms but may also affect non-target flora and fauna for long periods of time.

DDT and Metabolites: It is well documented that DDT and its metabolites, DDD and DDE, are concentrated by aquatic organisms from water and then are bioaccumulated by other organisms at higher trophic levels. Long term dietary dosage at 2.8 to 3 mg/kg DDE (wet weight) resulted in adverse reproductive effects in mallards (*Anas platyrhyncus*)(Heath, et al.,1969; Hazeltine, et al.,1974), black ducks (*Anas rubripes*)(Longcore et al.,1971; Longcore and Stendel, 1977), and screech owl (*Otus asio*)(McLane and Hall, 1972). Blus *et al.* (1974) reported normal reproductive success of brown pelican (*Pelecanus occidentalis*) eggs at concentrations <2.5 mg/kg DDT and its metabolites while Anderson (1975) reported crushed brown pelican eggs at average concentrations of 907 mg/kg DDT and its metabolites. Reproductive problems were observed in snowy egret (*Egretta thula*) eggs with DDE concentrations above 5 ppm and in night heron (*Nycticorax nycticorax*) eggs with concentrations above 8 ppm DDE (Henny et al.,1985). White-faced ibis (*Plegadis chihi*) eggs with DDE concentrations of 3 ppm or more cracked readily (Steele, 1984; Henny et al.,1985)

Tomatis, *et al.* (1971) reported the highest concentrations of DDT and its metabolites in fat tissue, reproductive organs, liver, kidneys, and the brain, in decreasing order, in laboratory mice (*Mus* spp.). Acute toxic effects of DDT are to the central nervous system with symptoms such as hyperexcitability, trembling, convulsions, and paralysis. The most consistent finding in lifetime feeding studies has been an increase in the size of the liver, kidneys and spleen, extensive degenerative changes in the liver and an increase in mortality rate. DDT's oral LD50 for rats (*Rattus* spp.) was reported at 113 mg/kg and 118 mg/kg (Gaines (1969) and Verschueren (1983) respectively). Oral LD50 for mice is 135 mg/kg, 250 mg/kg for rabbits (Lewis, 1992), and 60 mg/kg for dogs (*Canis* spp.)(Pimentel, 1971).

DDT and its metabolites DDD and DDE are known for their effects on piscivorous birds. American kestrels and pelicans experience reduced survival or reproduction at 3 mg/kg and 0.15 mg/kg in diet, respectively (Anderson, 1975; Lincer, 1975). The acute oral LD50 for birds is approximately 1000 mg/kg (Matsumura, 1985). Fish experience effects at 3-11 mg/kg body burden (USEPA, 1980c) and 3 mg/kg is lethal to cutthroat trout fry.

Dieldrin: Dieldrin is a manmade, chlorinated cyclic hydrocarbon insecticide compound, in a group of compounds which includes DDT and BHC. Dieldrin is persistent in the environment due to its extremely low volatility and low solubility in water. Dieldrin and other organochlorine pesticides have been found in

higher concentrations in addled (non-viable) than in viable eggs of several species of birds, including great horned owls and red-tailed hawks, and may cause eggshell thinning (Springer, 1980). Additionally, dieldrin is extremely polar and is retained in plant waxes and animal fats, leading to accumulation in the food chain (Sittig, 1985).

5.2.4 Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are moderately persistent in the environment, and may potentially cause adverse effects to vegetation, fish, and wildlife. A variety of adverse biological effects have been reported in numerous species of organisms under laboratory conditions, including carcinogenic effects, as well as effects on survival, growth, and metabolism (Eisler, 1987).

Toxic effects of the various PAHs differ among compounds, generally as a function of molecular weight. Unsubstituted lower molecular weight compounds containing two-three rings (e.g., naphthalene) exhibit acute toxicity and other adverse effects to some organisms, but are noncarcinogenic.

The potential effects of PAHs on aquatic biota include reduced survival, decreased food uptake, carcinogenesis, inhibited reproduction, decreased heart rate and respiration, increased weight of body organs in fish, and photosynthetic inhibition in algae and macrophytes (Eisler, 1987). PAHs vary substantially in their toxicity to aquatic organisms. In general, toxicity increases as molecular weight increases, although high molecular weight PAHs have low acute toxicity, perhaps due to their low solubility in water (Eisler, 1987). Many PAHs, especially lower molecular weight compounds, are acutely toxic at concentrations between 50 and 1,000 µg/l and sublethal effects have been documented at concentrations as low as 0.1 to 5.0 µg/l (Eisler, 1987). LOEL values for acute toxicity are 1,700 µg/l for acenaphthene (520 µg/l for chronic toxicity), 3,980 µg/l for fluoranthene, and 2,300 for naphthalene (620 µg/l for chronic toxicity) (USEPA, 1986).

Most species of aquatic organisms studied to date accumulate PAHs from low concentrations in the ambient medium (water and sediment). Uptake is highly species specific, being highest in species incapable of metabolizing PAHs. Bioaccumulation factors tend to increase as the molecular weight of the PAH compound increases, with increases in the amount of organic matter in the medium, and with increases in the lipid content of the organism (Eisler, 1987). Depuration rates vary by species, but are usually rapid, except in some species of invertebrates (Eisler, 1987). The role of sediment in PAH uptake can be important. When sediment PAH levels are elevated, benthos obtain a majority of their PAHs from sediments through their ability to mobilize PAHs from the sediment/pore water matrix (Eisler, 1987). The elevated levels in the tissues of these organisms could provide a significant source of PAHs to predatory fish. However, fish have the ability to efficiently metabolize and degrade PAHs (Eisler, 1987).

Brown bullheads, in response to repeated applications of Buffalo River sediment extracts, showed higher frequencies of epidermal tumors when compared to controls (Eisler, 1987). In a separate study, a positive relationship was established between sediment PAH levels and liver tumors in fish from the Black River, Ohio. Sediment PAH concentrations ranged from 50 to 100 mg/kg for some individual compounds. Brown bullheads exposed to the sediment contained from 1.1 to 5.7 mg/kg of several PAH compounds in their

tissues and exhibited a 33 percent higher frequency of liver tumors than controls (Eisler, 1987). In a third study, from the Niagara River in New York, brown bullheads had significantly higher total lesion incidences at a site heavily contaminated with PAHs, when compared with a reference site (Hickey et al.,1990).

Only limited data are available on the potential effects of PAHs on amphibians and reptiles. In amphibians and reptiles, as in mammals, the mixed-function of oxidase system acts to detoxify PAHs, although the rate of metabolism tends to be slower than in mammals. However, amphibians are quite resistant to PAH carcinogenesis, when compared to mammals (Eisler, 1987).

Little information on the toxicity of PAHs to birds has been collected. Two studies have been conducted on the toxicity of PAHs to mallards. When fed 4,000 mg total PAH (mostly as lower molecular weight compounds) per kg body weight for seven months, no mortality or visible signs of toxicity resulted. Other effects were noted, however, including an average increase in liver size of 25 percent, and increased blood flow to the liver of 30 percent. In the second study, adverse sublethal effects were noted at concentrations of between 0.036 and 0.18 µg PAH per egg following application of various PAHs (e.g., chrysene and benzo(a)pyrene) to the surface of mallard eggs (Eisler, 1987). It has been suggested that the presence of PAHs in petroleum may confer many of the well-documented adverse biological effects reported after eggs have been exposed to polluting oils (Albers and Gay, 1982; Hoffman and Albers, 1984).

Numerous studies have shown that unsubstituted PAHs do not accumulate in mammalian adipose tissue, despite their high lipid solubility, probably because they tend to be rapidly and extensively metabolized (Eisler, 1987). Thus, long-term storage and biomagnification through food chains is not likely to occur for PAHs. Acute oral LD50s for rats range from 50 to 9430 mg/kg with a median of approximately 1000 mg/kg (USEPA, 1980d; Eisler, 1987; NIOSH, 1988).

5.2.5 Polychlorinated Biphenyls (PCBs)

PCBs are persistent, bioaccumulative, and highly toxic. Mink are the most sensitive species to PCBs, experiencing reproductive failure at 0.64 mg/kg in diet (Ringer, 1983; Fuller and Hobson, 1986). Birds experience reproductive and immunotoxic effects at 10-40 mg/kg in diet (Peakall, 1986). Because these levels induce catastrophic reproductive effects, the International Joint Commission (IJC, 1988) recommends a concentration in fish of 0.1 mg/kg to protect piscivores. The acute dietary LC50 in birds is 747-12,000 mg/kg (Peakall, 1986).

5.2.6 Phthalate Esters

Oral LD50 values for mammals range from 1000 to 34,000 mg/kg, with a median across esters and species of approximately 10,000 mg/kg (EPA 1980f, NIOSH 1988). Threshold dietary effects levels in rats are 40,000 mg/kg for di-n-butylphthalate and 2000 mg/kg for bis(2-ethylhexyl) phthalate (USEPA, 1980f; NIOSH, 1988).

5.2.7 Dioxins/Furans

2,3,7,8-TCDD: A number of toxic responses have been observed following exposure to 2,3,7,8-TCDD; responses are marked by interspecies variability, with some responses being highly species specific and

confined to one or a few species. Loss of body weight or reduced weight gain and thymic atrophy are the most consistent toxic responses of 2,3,7,8-TCDD exposure in various species, with the latter being one of the most sensitive indicators of toxicity. In general, the toxicologic pattern observed with 2,3,7,8-TCDD is not unique; it also occurs with certain halogenated dibenzofurans, chlorinated biphenyls, naphthalenes, and brominated dioxins (McConnell, 1980).

Data published by Miller et al. (1973) and Norris and Miller (1974) indicated that the 96-hour LC50s for a worm, *Paranais* sp., a snail, *Physa* sp., and larvae of the mosquito, *Aedes aegypti*, would be >0.2 µg/l, whereas those for the coho salmon, *Oncorhynchus kisutch*, and the guppy, *Poecilia reticulata*, would be >1 and >10 µg/l, respectively. Based on microcosm studies in which concentrations in water were measured at 2-day intervals, the 96 hours LC50 for fingerling channel catfish, *Ictalurus punctatus*, would be >0.24 µg/l, whereas those for *Daphnia magna* and a snail, *Physa* sp., would be >1.3 µg/l (Isensee and Jones, 1975; Isensee, 1978). Yockim et al. (1978) did not observe acute toxicity to *D. magna*, a snail, *Helosoma* sp., or the mosquitofish, *Gambusia affinis*, exposed for over 96 hours to a measured concentration of 0.0024-0.0042 µg/l. Helder (1980, 1981, 1982) found that the 96-hour LC50s for embryos of northern pike, *Esox lucius*, and embryos and yolk-sac fry to rainbow trout, *Salmo gairdneri*, would be >0.01 µg/l; the 96-hour LC50 for juvenile rainbow trout would be >0.1 µg/l. Although no 48- or 96-hour LC50s or EC50s can be calculated, the available data indicate that those for the coho salmon, guppy, *D. magna*, and a snail, *Physa* sp., are >1.0 µg/l.

Because Miller et al. (1973) used static long-term exposures, no conclusions can be drawn concerning chronic toxicity from their exposures of *A. aegypti* or a snail, *Physa* sp., but it can be concluded that 0.2 µg/l would cause chronic toxicity to a worm, *Paranais* sp. A 96-hour exposure to an initial concentration of 0.0056 µg/l resulted in 55% mortality among coho salmon within 60 days (Miller et al.,1973, 1979); thus 0.0056 µg/l would cause chronic toxicity to this species. Similarly, 0.1 µg/l would cause chronic toxicity to the guppy, because exposure to 0.1 µg/l for 5 days killed all individuals within 40 days (Norris and Miller, 1974). In microcosms in which the concentrations of 2,3,7,8-TCDD were measured at 2-day intervals, both *D. magna* and a snail, *Physa* sp., reproduced at 1.3 µg/l (Isensee and Jones, 1975; Isensee, 1978). Exposure to a measured concentration of 0.0024-0.0042 µg/l killed all exposed mosquitofish and channel catfish within 20 days (Yockim et al.,1978). Based on effects caused by 96-hour exposures, 0.001 µg/l would cause chronic toxicity to rainbow trout and 0.01 µg/l would chronically affect northern pike, Helder, 1980, 1981, 1982). Branson et al. (1983) reported that a 6-hour exposure to 0.1 µg/l adversely affected rainbow trout after 64-139 days. Apparently 0.001 µg of 2,3,7,8-TCDD/l would cause unacceptable chronic toxicity to rainbow trout and 0.01 µg/l would be chronically toxic to coho salmon, mosquitofish, channel catfish and northern pike; 1.3 µg/l may not be chronically toxic to *D. magna* or a snail, *Physa* sp.

Several measured BCFs have been reported for 2,3,7,8-TCDD. Using microcosm studies in which the concentrations in water were measured at 2-day intervals for 30-33 days, Isensee and Jones (1975) and Isensee (1978) obtained BCFs of 390-13,000 for the alga, *O. cardiacum*, a snail, *Physa* sp., and *D. magna*. In a separate 32-day microcosm study in which the measured concentrations of 2,3,7,8-TCDD ranged from 0.0024-0.0042 µg/l, BCFs for *O. cardiacum*, *Physa* sp., and *D. magna* ranged from 660-7070 from the seventh day to the end of the test.

The oral LD50 values range from 0.6 µg/kg body weight for guinea pigs to 5.05 mg/kg bw for hamsters (Schwetz et al. 1973; Vos et al. 1974; McConnell et al. 1978a,b; Henck et al.,1981; Olson et al.,1980). The dermal LD50 for rabbits was 275 µg/kg of body weight (Schwetz et al.,1973); death was sometimes delayed as long as 40 days following acute exposure. Of the laboratory animals studied, the guinea pig was the most susceptible to the toxic effects of 2,3,7,8-TCDD (Schwetz et al.,1973; Gupta et al.,1973; Greig et al.,1973).

6.0 RISK CHARACTERIZATION

Risk characterization quantitatively defines the magnitude of potential risks to ecological receptors under a specific set of circumstances. It is the process of applying numerical methods and professional judgement to determine whether adverse effects are occurring or are likely to occur due to the presence of COCs at a given study site. Risk characterization should address the following questions: (a) Are ecological receptors currently exposed to site-related stressors at levels capable of causing harm, or is future exposure likely?, (b) If adverse ecological effects are observed or predicted, what are the types, extent, and severity of effects? and, (c) What are the principle uncertainties associated with the risk characterization?

6.1 Risk Estimation

For this screening-level assessment, risk estimation involved a quantitative comparison of estimated exposure point concentrations with TRVs for invertebrate and vertebrate receptors to identify the potential for occurrence of adverse effects due to direct (invertebrates and fish in water) and secondary (consumption of contaminated prey by piscivores) exposures. A Toxicity Quotient (TQ) was calculated to facilitate this comparison, as follows:

$$TQ_{km} = \log(EPC_k / TRV_m) \qquad \text{[Eq. 6-1]}$$

where: TQ_{km} = ecological quotient for the *m*th indicator species relative to the *k*th COC, EPC_k = exposure point concentration (from Table 4-1) for the *k*th indicator species, and TRV_m = toxicity reference value (from Table 5-1) for the *m*th COC. TQ values are shown in Table 6-1, where positive values indicate that exposure is greater than acceptable levels, negative values indicate that it is not, and zero indicates that exposure equals acceptable levels. Each whole integer above zero indicates an order of magnitude increase in the potential for toxic effects. For example, a TQ of 2 shows that the exposure point concentration is approximately equal to the LD50 and that severe effects are possible in 50% of the population. An TQ is not a direct measure of risk, but merely a convenient method for indicating exceedence of acceptable values.

This quotient method will tend to over-estimate the potential for adverse impacts, because: (a) factors, such as bioavailability from sediment or surface water, degradation rates in sediment or surface water, metabolic transformation in vegetation or invertebrates, receptor avoidance of contaminated sediments or surface water, dilution over distance, or frequency of receptor exposure to contaminated media, that might reduce exposure values are not considered; (b) ecological receptor home and foraging ranges are assumed to be completely within the contaminated areas of the river, causing receptors to be exposed at the upper range of media concentrations at all times, which is not likely the case; (c) estimates of COC concentrations in prey and forage items do not take into account either absorption or elimination factors that, if applied, could significantly reduce estimated tissue residue levels.

6.2 Risk Description

Risk description involves summarizing and interpreting the ecological significance of any observed or predicted effects and the degree of risk they pose to ecological receptors. Interpretation of ecological significance must take into account such factors as nature and magnitude of effects, spatial and temporal

TABLE 6-1
Sediment Toxicity Quotient (TQ) Values by Reach for Ecological Receptors

CONSTITUENT OF CONCERN	TQ VALUES FOR INVERTEBRATE RECEPTOR			TQ VALUES FOR FISH RECEPTOR			TQ VALUES FOR PISCIVOROUS RECEPTOR		
	upstream	facility	downstream	upstream	facility	downstream	upstream	facility	downstream
INORGANICS									
Arsenic	-0.35	-0.02	-0.56	-0.28	0.05	-0.49	-1.61	-1.29	-1.82
Barium	1.10	1.76	1.09	1.10	1.76	1.09	- dg -	- dg -	- dg -
Chromium	-0.56	0.85	-0.63	-0.70	0.71	-0.77	-3.43	-2.03	-3.51
Copper	2.94	3.98	2.55	3.94	4.98	3.55	-0.96	0.09	-1.34
Cyanide		- dg -			- dg -			- dg -	
Lead	-0.52	0.16	-0.65	-0.09	0.59	-0.22	-2.13	-1.45	-2.26
Mercury	0.83	2.15	0.49	0.65	1.97	0.31	-0.10	1.22	-0.44
Nickel	-0.78	-0.08	-1.16	0.85	1.55	0.47	-2.28	-1.58	-1.66
Silver		2.64			2.56			-1.54	
Tin		-0.16			1.00			-2.16	
Vanadium	-0.93	-0.40	-1.24	-0.74	-0.21	-1.06	-2.79	-2.26	-3.11
Zinc	1.72	3.50	1.58	1.72	3.50	1.58	-0.27	1.52	-0.41
ORGANICS									
1,2-Dichlorobenzene	-0.54	-0.50		-1.05	-1.02		-2.10	-2.06	
1,4-Dichlorobenzene		-1.60			-0.04			-1.18	
Chlorobenzene	0.81	0.54	0.48	-0.33	-0.60	-0.66	-2.17	-2.44	-2.50
Tinuvin 328		- dg -	- dg -		- dg -	- dg -		- dg -	- dg -
Toluene	0.27	3.34	0.72	-1.58	1.49	-1.13	-4.25	-1.17	-3.80
Xylene (m & p)	-1.29	-0.81		-1.91	-1.43		-3.71	-3.23	
Xylene (o)	-0.77	-1.14		-1.39	-1.76		-3.18	-3.56	
PESTICIDES									
2,4-D		1.77			-0.34			-1.63	
4,4'-DDE		-1.68	-1.26		-3.22	-2.79		-4.70	-4.28
4,4-DDT	-1.92	-1.74	-1.50	-1.28	-1.10	-0.86	-1.38	-1.20	-0.96
BHC, alpha-		0.25			-0.49			-4.18	
Chlordane, gamma-	-1.24	-1.79	-1.52	-1.82	-2.37	-2.10	-2.45	-3.00	-2.73
Dieldrin		0.01	0.71		0.21	0.91		-2.56	-1.86
Disulfoton		-0.05	0.50		-0.52	0.02		-2.10	-1.56
Endrin		-3.38			-1.80			-3.55	
Heptachlor	0.12	-0.24	0.54	-0.57	-0.93	-0.14	-2.32	-2.68	-1.90
Pentachlorophenol		-0.21			0.45			-2.63	

TABLE 6-1

Sediment Toxicity Quotient (TQ) Values by Reach for Ecological Receptors

CONSTITUENT OF CONCERN	TQ VALUES FOR INVERTEBRATE RECEPTOR			TQ VALUES FOR FISH RECEPTOR			TQ VALUES FOR PISCIVOROUS RECEPTOR		
	upstream	facility	downstream	upstream	facility	downstream	upstream	facility	downstream
PAHs									
Acenaphthylene		-4.60			-3.36			-2.14	
Anthracene	-0.24	-0.61	-0.01	1.13	0.76	1.36	-1.29	-1.66	-1.07
Benzo(a)anthracene	-0.13	-0.67	-0.03	2.25	1.71	2.35	-0.44	-0.97	-0.34
Benzo(a)pyrene	-0.47	-1.00	-0.39	1.89	1.37	1.97	-0.57	-1.10	-0.49
Benzo(b)fluoranthene	- tv -	- tv -	- tv -	- tv -	- tv -	- tv -	-0.36	-0.82	-0.31
Benzo(g,h,i)perylene	-0.16	-0.66	-0.08	- tv -	- tv -	- tv -	-0.74	-1.24	-0.66
Benzo(k)fluoranthene	-0.34	-0.83	-0.28	- tv -	- tv -	- tv -	-0.37	-0.86	-0.31
Chrysene	0.99	0.56	1.23	- tv -	- tv -	- tv -	-0.29	-0.72	-0.05
Dibenzofuran	-0.31	-0.78	-0.09	-0.31	-0.78	-0.09	-1.57	-2.03	-1.35
Fluoranthene	-0.21	-0.72	0.05	-2.12	-2.63	-1.85	0.16	-0.35	0.42
Fluorene	-0.53	-0.77	-0.31	-0.85	-1.10	-0.64	-1.09	-1.34	-0.88
Indeno(1,2,3-cd)pyrene	- tv -	- tv -	- tv -	-5.07	-5.52	-4.96	-0.92	-1.38	-0.82
Naphthalene	-1.31	-0.70	-0.44	-0.76	-0.14	0.11	-2.05	-1.43	-1.18
Phenanthrene	-1.52	-1.88	-1.32	-2.40	-2.75	-2.19	-0.21	-0.56	0.00
Pyrene	- tv -	- tv -	- tv -	1.81	1.45	1.98	-0.21	-0.57	-0.05
PCBs									
PCB-1248		-0.22			-2.65			-0.92	
PCB-1254		-1.99			-3.12			-3.21	
PHTHALATE ESTERS									
Di-n-butylphthalate	-4.49	-3.74	-3.81	-4.49	-3.74	-3.81	-2.21	-1.46	-1.54
Di-n-octylphthalate		- tv -	- tv -		- tv -	- tv -		- tv -	- tv -
DIOXINS/FURANS									
HxCDD		-1.97			-0.07			-0.82	
HxCDF		-2.07			-0.17			-0.92	
PeCDD		-3.02			-1.13			-1.87	
PeCDF		-2.46			-0.57			-1.31	
TCDD		-3.08			-1.18			-1.93	
TCDF		-3.09	-2.33		-1.19	-0.44		-1.94	-1.18

"- dg - " = data gap in physiochemical data; parameter required to calculate exposure value not available, see Table 4-1

blank = COC not detected in environmental media (Table 3-2)

"- tv -" = data gap in toxicity data; toxicity reference value not available, see Table 5-1

distribution of effects, and the potential for study site recovery. A weight-of-evidence approach is used wherein several qualitative and quantitative lines of evidence are integrated to describe ecological risks posed by the Facility.

Results obtained from this screening-level assessment provide preliminary answers to the questions of interest to this screening assessment (c.f., Problem Formulation, Section 3.2), as follows:

- ***Are ecological receptors currently exposed to site-related COCs at levels capable of causing harm?***

  Ecotoxicological analysis (Tables 6-1 and 6-2) indicates a somewhat greater potential for adverse effects at Facility stations than at either upstream or downstream stations. Barium, copper, mercury (inorganic), toluene, silver, benzo(a)anthracene, and zinc, present significant risks to invertebrates and fish receptors within the Facility reach. However, some of these COCs (notably barium, copper, toluene, zinc) are also present at significant concentrations at upstream stations, which suggests a potential for adverse effects at these stations, as well as a source contributing to adverse effects within the Facility.

  Barium, copper, and zinc at upstream stations are present at levels that could produce adverse acute effects in 25% to >50% of individuals comprising invertebrate and fish populations, as well as severely limit recruitment of more sensitive species. Risks associated with these COCs remain a significant threat throughout the length investigated. While PAHs present some degree of risk to fish in all reaches, they were never specifically present in the Facility waste streams and could have been released from a variety of non-point sources unrelated to the Facility. The majority of COCs examined do not appear to present significant risks to piscivorous avian receptors. Only copper, mercury, and zinc might pose potential risks in the Facility reach.

- ***If adverse ecological effects are observed or predicted, what are the types, extent, and severity of effects?***

  Benthic and fish community data indicate that some adverse impacts are occurring, but not necessarily in any specific relationship to the Facility reach. Preliminary analysis of spring (June 1993) benthic sampling data shows stations adjacent to the Facility to be not distinctly different in terms of species richness, dominance, or diversity when compared to upstream (reference) or downstream stations. In addition, there is no statistically significant ($r^2 < 0.20$) trend in these parameters along the river gradient. Tubifex worms, a pollution-tolerant species, were the numerically dominant species at every station. These findings suggest that the Pawtuxet River benthos is under some degree of chronic stress throughout the length investigated.

  Fish surveys show a community dominated by suckers and carp; more sensitive species were either few in number or absent from the samples collected. Based on an initial estimate of population age structures, normal recruitment appears to be occurring in white sucker populations but not in carp populations. Although carp are relatively pollution-tolerant species, this could indicate reproductive impairment in carp due to the presence of contaminants. However, it is

Table 6-2
Summary of Toxicity Quotient Results by Reach and Receptor

Toxicity Quotient Range	INVERTEBRATES			FISH			PISCIVORES		
	upstream stations	facility stations	downstream stations	upstream stations	facility stations	downstream stations	upstream stations	facility stations	downstream stations
4.00 - 4.99					copper				
3.00 - 3.99		copper zinc toluene		copper	zinc toluene	copper			
2.00 - 2.99	copper	mercury silver	copper	benzo(a)anthracene	silver	benzo(a)anthracene			
1.00 - 1.99	barium zinc	barium 2,4-D	barium zinc chrysene	barium zinc anthracene benzo(a)pyrene pyrene	barium mercury nickel toluene benzo(a)anthracene benzo(a)pyrene tin pyrene	barium zinc anthracene benzo(a)pyrene pyrene		mercury zinc	
0.00 - 0.99	mercury toluene chrysene chlorobenzene heptachlor	chromium lead alpha-BHC dieldrin chrysene chlorobenzene	mercury toluene dieldrin disulfoton heptachlor chlorobenzene fluoranthene	mercury nickel	arsenic chromium lead dieldrin anthracene	mercury nickel dieldrin disulfoton naphthalene	fluoranthene	copper	fluoranthene

0 → NOEL (NO ADVERSE EFFECT)

equally plausible that: (a) the carp breed and reproduce in areas of the river outside the length investigated or (b) that the younger cohorts in the carp population are subject to intense predation pressure from larger carp who are voracious, indiscriminant feeders.

Abnormalities are present on fish collected throughout the area and, although the exact cause of these pathologies is currently unknown, they are of a type frequently associated with chemical pollution. Although carp have been reported to return to the same breeding area year after year, it is not known whether either carp or white suckers exhibit a strong home range fidelity or can move freely along the length of the river as mill dams may limit their movement at normal river flow levels. As with any potentially highly mobile species, it is not possible to suggest a strict cause-and-effect relationship between dermal abnormalities observed in these fish and the presence of contaminants within the Facility reach. For the same reason, chemical analysis of fish tissue, even if COCs are detected, is likely to be inconclusive in linking specific COCs with observed abnormalities.

Although COCs were present in surface water, bioassay testing was unable to identify significant mortality in invertebrate or fish test species exposed to surface water samples. This suggests that surface water COCs, while detectable, are not necessarily bioavailable to ecological receptors. Exposure to both pore water and sediment induced significant levels of mortality in test species, indicating that sediments are most likely the primary sources of toxicity. While mortality was generally greatest in sediments sampled at the Facility, some instances of mortality were measured in sediments and pore water collected both upstream and downstream. Mortality in upstream samples suggests a possible source contributing to adverse effects at the Facility. Mortality in downstream sediment samples indicates either significant downstream transport of Facility reach sediments or another significant contaminant source downstream.

- ***To what extent do contaminants present in the upstream reach contribute to the potential for adverse impacts within the Facility reach?***

  The TQ values shown in Table 6-1 were used to group COCs based on the reach or reaches in which they were estimated to induce toxic effects in any receptor (Table 6-3). Several COCs (barium, copper, mercury, nickel, zinc, chlorobenzene, toluene, heptachlor, and some PAHs) are present at toxic levels in upstream reach sediments and any downstream migration on their part may contribute to toxicity in the Facility and downstream reaches.

- ***Are there contaminants whose potential for adverse impacts is confined to the Facility reach?***

  Overall, 22 of 28 COCs displaying significantly elevated TQ values are present in the Facility reach and arsenic, silver, tin, 2,4-D, and α-BHC exhibit potential toxic effects exclusively within this reach. Some otherwise widely distributed COCs, specifically barium, copper, toluene, and zinc, exhibit their highest TQ values within the Facility reach, which suggests either a significant source, or hydrologic conditions that caused their retention, within this reach. Upstream discharges by waste water treatment plants and industrial facilities must be seen as confounding factors when

attempting to identify and isolate Facility-related impacts to the river ecosystem.

- ***To what extent do contaminants present in the Facility reach contribute to the potential for adverse impacts within the downstream reach?***
  Several COCs (fluoranthene, naphthalene, and phenanthrene) are present only in the downstream reach at potentially toxic concentrations. These COCs could have originated in the Facility reach and been deposited in the downstream reach; however, because these are either pesticides or PAHs, they could also have reached the river from any one of several non-point sources, such as stormwater runoff or atmospheric deposition. Although the benthic invertebrate and bioassay data hint at the potential for other contaminant sources downstream, this screening ecotoxicological analysis cannot define a significant, unique downstream source.

6.3 Uncertainty Analysis

To ensure that ecological receptors are protected during a screening-level assessment, numerous assumptions are made that tend to overestimate rather than underestimate potential risks. The above conclusions are based on the data and assumptions specified. These conclusions should be judged as conservative, since overriding uncertainties associated with estimating impacts that may result from any COC exposure include the following:

(a) Factors that might reduce exposure values, such as bioavailability from soil or surface water, degradation rates in soil or surface water, metabolic transformation in vegetation or invertebrates, receptor avoidance of contaminated soils or surface water, dilution over distance, or frequency of receptor exposure to contaminated media, were not factored into this screening analysis. These conditions are expected to create an over-estimation of COC exposure concentrations.

(b) Applicability of literature-derived data depends upon types of results presented and methods used to arrive at these results. Test endpoints produced by laboratory and field tests may be reported as formally defined toxicological endpoints or as less stringently defined measures of mortality or sublethal effect; variations in format introduce a source of error when subsumed into a single TRV. Thus seemingly equivalent TRVs may be significantly different owing to differences in test protocols, test conditions, or responses of individual organisms (Lewis et al., 1990).

(c) Terrestrial and aquatic species home ranges, and therefore exposures, are assumed to be completely within the contaminated areas of each study site. Thus, plants and animals are assumed to be exposed at the upper range of media concentrations at all times, which is not likely the case.

(d) Impacts to individual organisms are considered in this screening assessment, rather than impacts to populations. Generally, except for threatened and endangered species, assessments need only to evaluate population effects (USEPA, 1989a). Evaluating risks to individual organisms tends to overestimate risks to both populations and communities.

(e) Estimates of COC concentrations in prey and forage items are based on the simplest possible equilibrium partitioning model, which does not take into account either absorption or elimination factors and which could cause an overestimate of tissue residue levels and thus of risk.

(f) Regulatory standards, criteria, and/or toxicological data were not available for every COC and thus

they could not be evaluated for potential impacts. Data gaps may cause an underestimate of risk because unevaluated COCs could be unrecognized sources of risk. Conversely, substitutions of similar compounds, such as hexachlorobenzene for chlorobenzene, tend to overestimate risk.

(g) The COCs identified in this report may not be the only, or even the most significant, sources of stress to the Pawtuxet River over the length investigated. Conditions such as chlorine in effluents from waste water treatment facilities, oxygen deficient conditions, increases in water temperture, low water levels and flows during summer months, and eutrophication, may all contribute to the stresses endured by ecological receptors in the Pawtuxet River.

(h) Ecotoxicological chronic effects data (NOAEL) were not available for use as TRVs for every COC and acceptable TRVs had to be linearly extrapolated from actute effects data (LOAEL, LD50, LC50, EC50) through the use of uncertainty factors. The assumptions inherent in the selection and use of these factors can be a source of uncertainty and will, if the factors are large, provide an overestimate risk. Comparison of TRVs to RME concentrations, represented by the upper 95% confidence interval of the mean, is an inherently conservative method which will tend to overestimate both exposure levels and risk. This source of uncertainty is also significant when extrapolating from acute to chronic bioassay data.

6.4 Conclusions & Recommendations

Results produced by this screening assessment based on field surveys, bioassay tests, and simple ecotoxicological models suggest that conditions along the length of the Pawtuxet River investigated do not meet the assessment endpoint for aquatic species (assessment endpoint (a), Table 3-4), as COCs are present in the Pawtuxet River ecosystem at concentrations potentially capable of adversely impacting benthic organisms and fish. Some degree of chronic stress, most probably from chemical stressors, is evident in benthic invertebrate and fish populations throughout the length investigated. Although the reach adjacent to the Facility may contribute to these observed and estimated adverse effects, it is clearly not the only stressor source nor is it necessarily the most significant. A few widely distributed, highly toxic, and non-Facility specific COCs (most notably copper and zinc) are undoubtedly responsible for at least some of the ecological stress observed in benthos and fish throughout the length investigated.

Ecotoxicological results produced by this screening assessment suggest that the Pawtuxet River has a high probability of meeting the assessment endpoints for wildlife species (assessment endpoints (b & c), Table 3-4) because the potential for adverse impacts in terrestrial, piscivorous species from the consumption of COCs bioaccumulated in fish prey was estimated to be minimal. This position may be substantiated with more detailed food web modeling in the baseline ecological risk assessment.

Ecological values in the Pawtuxet River worthy of preservation or restoration could include a healthy, functioning benthic infauna and fish populations with normal demographic characteristics. Remedial actions taken to address site-related contaminants in the Facility reach would contribute to the restoration of better ecological values in the Pawtuxet. However, the river ecosystem is unlikely to receive maximum benefits from any actions unless contaminant sources not related to the Facility are also addressed. These include: (a) contaminated sediments upstream of the Facility reach, (b) waste water treatment plant and industrial discharges upstream of the Facility reach, and (c) non-point source discharges, such as storm

runoff and atmospheric deposition, that enter the river at numerous points along the length investigated.

Based on the results of this screening assessment, it is recommended that the baseline ecological risk assessment for the Pawtuxet River focus on a group of eight "indicator" COCs selected from the COCs identified in Table 6-1. This will permit a more thorough examination of COCs that either make the greatest contribution to the overall potential for toxic effects in the river or are more clearly Facility-related or both. The following list of "indicator" COCs is recommended:

- ***Copper:*** This is estimated to be the single greatest contributor to toxic effects along the length investigated and specifically, with a TQ of 4.98, within the Facility reach. Copper is seen as a substantial contributor to total contaminant loading. Although they exhibit high TQ values, barium and mercury were not selected as indicators because it is thought that they are not present in a soluble, bioavailable form.
- ***Chlorobenzene:*** This is the only benzene-related compound to exhibit the potential for adverse effects within every reach. It is also thought to be historically associated with Facility processes.
- ***Naphthalene:*** This is a representative PAH that is thought, unlike many of the other PAHs have been used as a raw material. However, it is not present in concentrations great enough to induce toxic effects, except in the downstream reach (TQ = 0.11).
- ***PCB-1254:*** Despite the fact that it yielded no TQ greater than zero, it is a highly toxic, lipophillic, bioaccumulative COC whose presence should be monitored.
- ***Tinuvin 328:*** This organic compound was selected because it is unique to processes at the Facility and can be clearly related to Facility operations. It has a relatively low toxicity but could serve as one "marker" for the extent of Facility related contamination.
- ***Toluene:*** This organic compound is associated with Facility operations and is thought to be entering the river within the Facility reach as a result of groundwater discharges from the Site. It is estimated to present a significant risk to invertebrate receptors (TQ = 3.34).
- ***Silver:*** This COC was only detected in Facility reach sediments and poses a significant (TQ > 2.50) threat to both invertebrate and fish receptors.
- ***Zinc:*** After copper, this COC, with a TQ of 3.50, is the second greatest contributor to toxic effects within the Facility reach. It is also a COC that is definitely known to be a major component of historic Facility waste streams. When the fact that zinc may be regulated at the cellular level is overlooked, it is estimated to present a risk to piscivorous receptors.

Revised toxicity reference values for these indicator COCs, supported by detailed toxicity data, are provided in Appendix A. These revised values may be used to support both a baseline ecological risk asssessment and establishment of ecological media protection standards (MPS) for the Pawtuxet River. Pesticides, phthalate esters, dioxins, furans, and PAHs (other than naphthalene) were not identified as indicator COCs because they were not estimated to contribute significantly to toxicity, were not historically associated with Facility related waste streams, and could have emanated from a number of sources unrelated to the Facility.

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APPENDIX A

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TABLE A-1
CIBA Cranston Facility
Comparison of Toxicity Values for Aquatic Receptors

Indicator Constituent of Concern	Final Chronic Value[d] (mg/L)	Toxicity Reference Value[f]		Lowest Chronic Value[c]	
		fish (mg/L)	invertebrates (mg/L)	fish (mg/L)	daphnids (mg/L)
copper	0.012[b]	0.038	0.00023	0.003873	0.00023
silver	0.00012[b]	0.00011	0.0028	0.00012	0.0026
zinc	0.11[b]	0.0173	0.0151	0.03641	0.04673
chlorobenzene	0.0116[a]	0.245	0.173	1.203	15.042
naphthalene	0.0021[a,h]	0.056	0.0396	0.62	1.163
PCB-1254	0.00002[a]	0.0036[g]	0.0009[g]	0.001	0.0021
Tinuvin 328	---	~ 1.0[e]	~ 1.0[e]	---	---
toluene	0.0104[a]	0.3168	3.448	1.269	25.229

NOTES

[a] advisory value; from Suter et al., 1992
[b] EPA NAWQC value
[c] chronic value is geometric mean of the LOEC and NOEC; from Suter et al., 1992
[d] final acute value (FAV) / final acute-chronic ration (FACR); Stephan et al., 1985
[e] estimated as 0.01 x LC-50, using data reported by CIBA
[f] geometric mean of 24,48,72, and 96 hour LC-50 values in Table 2
[g] computed using all LC-50 values
[h] EPA reports a LOEL of 0.620 mg/L as the freshwater chronic criteria

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
FISH						
Copper	7440-50-8	*Lepomis macrochirus*	LC-50	96	1.1	Benoit, 1975
			LC-50	96	8.3	Gecker, et al., 1976
			LC-50	96	0.32	Oseid, and Smith, 1972
			LC-50	96	2.4	O'Hara, 1971
			LC-50	96	0.9	Thompson, et al., 1980
			LC-50	96	1.25	Cairns and Scheier, 1968
		Pimephales promelas	LC-50	96	2	Brungs, et al., 1976
			LC-50	96	3.5	Brungs, et al., 1976
			LC-50	96	16	Brungs, et al., 1976
			LC-50	96	20	Brungs, et al., 1976
			LC-50	96	2.2	Brungs, et al., 1976
			LC-50	96	2.8	Brungs, et al., 1976
			LC-50	96	1.6	Brungs, et al., 1976
			LC-50	96	11	Brungs, et al., 1976
			LC-50	96	12	Brungs, et al., 1976
			LC-50	96	9.7	Brungs, et al., 1976
			LC-50	96	21	Brungs, et al., 1976
			LC-50	96	5.6	Brungs, et al., 1976
			LC-50	96	3.3	Brungs, et al., 1976
			LC-50	96	5	Brungs, et al., 1976
			LC-50	168	0.07	Norberg and Mount, 1985
Silver	7440-22-4	*Lepomis macrochirus*	LC-50	96	0.013	Holcombe, et al., 1987
		Pimephales promelas	LC-50	96	0.009	Holcombe, et al., 1987
Zinc	7440-66-6	*Pimephales promelas*	LC-50	96	0.238	Norberg and Mount, 1985
			LC-50	168	0.238	Norberg and Mount, 1985
Zinc (II)[c]		*Lepomis macrochirus*	LC-50	96	4.7	LeBlanc, 1980a
		Pimephales promelas	LC-50	96	4.7	LeBlanc, 1980a
Chlorobenzene	108-90-7	*Lepomis macrochirus*	LC-50	24	17	Buccafusco, et al., 1981
			LC-50	96	16	Buccafusco, et al., 1981
			LC-50	48	16	Buccafusco, et al., 1981
			LC-50	24	24	Pickering and Henderson, 1966
			LC-50	24	17	Buccafusco, et al., 1981
			LC-50	48	24	Pickering and Henderson, 1966
			LC-50	96	24	Pickering and Henderson, 1966

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
			LC-50	96	.16	Buccafusco, et al., 1981
		Pimephales promelas	LC-50	24	29.12	Pickering and Henderson, 1966
			LC-50	24	33.93	Pickering and Henderson, 1966
			LC-50	24	39.19	Pickering and Henderson, 1966
			LC-50	48	34.98	Pickering and Henderson, 1966
			LC-50	48	33.93	Pickering and Henderson, 1966
			LC-50	48	29.12	Pickering and Henderson, 1966
			LC-50	96	29.12	Pickering and Henderson, 1966
			LC-50	96	33.93	Pickering and Henderson, 1966
			LC-50	96	22.2	Mayes, et al., 1983
			LC-50	96	22.3	Mayes, et al., 1983
			LC-50	96	35.4	Mayes, et al., 1983
			LC-50	96	16.9	Geiger, et al., 1990
Napthalene	91-20-3	*Pimephales promelas*	LC-50	24	7.76	Holcombe, et al., 1984
			LC-50	48	6.35	Holcombe, et al., 1984
			LC-50	72	6.08	Holcombe, et al., 1984
			LC-50	96	6.08	Holcombe, et al., 1984
			LC-50	96	7.9	Degraeve, 1982
			LC-50	96	6.14	Geiger, et al., 1985
			LC-50	96	1.99	Millemann, et al., 1984
PCB-1254	11097-69-1	*Lepomis macrochirus*	LC-50	96	2.74	Johnson and Finley, 1980
			LC-50	96	2.74	Stalling and Mayer, 1972; Johnson and Finley, 1980
			LC-50	240	0.443	Stalling and Mayer, 1972
			LC-50	360	0.204	Stalling and Mayer, 1972
			LC-50	360	0.303	Mayer, et al., 1977
			LC-50	480	260	Mayer, et al., 1977
			LC-50	480	0.135	Stalling and Mayer, 1972
			LC-50	600	0.054	Stalling and Mayer, 1972
			LC-50	600	0.239	Mayer, et al., 1977
			LC-50	720	0.177	Mayer, et al., 1977
			LC-50	96	2.74	USFWS, 1980
			LC-50	600	0.054	USFWS, 1986
		Pimephales promelas	LC-50	96	0.033	Nebeker, et al., 1974
			LC-50	96	0.0077	Nebeker, et al., 1974
Tinuvin 328		*Brachydanio rerio*	LC-50	96	100	Ciba Geigy, 1994

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
Toluene	108-82-1	*Lepomis macrochirus*	LC-50	24	109	Buccafusco, et al., 1981
			LC-50	48	3.4	Buccafusco, et al., 1981
			LC-50	96	3.02	Holcombe,et al., 1987
			LC-50	24	24	Pickering and Henderson., 1966
			LC-50	24	17	Buccafusco, et al, 1981
			LC-50	48	24	Pickering and Henderson., 1966
			LC-50	96	24	Pickering and Henderson., 1966
			LC-50	96	13	Buccafusco, et al., 1981
			LC-50	96	170	Johnson and Finley, 1980
		Pimephales promelas	LC-50	24	46.31	Pickering and Henderson, 1966
			LC-50	24	56	Pickering and Henderson., 1966
			LC-50	48	56	Pickering and Henderson., 1966
			LC-50	48	46.31	Pickering and Henderson., 1966
			LC-50	96	34.27	Pickering and Henderson., 1966
			LC-50	96	42.33	Pickering and Henderson., 1966
			LC-50	96	36	Devlin, et al., 1982
			LC-50	96	18	Devlin, et al., 1982
			LC-50	96	25	Devlin, et al., 1982
			LC-50	96	72	Devlin, et al., 1982
			LC-50	96	27	Devlin, et al., 1982
			LC-50	96	55	Devlin, et al., 1982
			LC-50	96	59	Devlin, et al., 1982
			LC-50	96	66	Devlin, et al., 1982
			LC-50	96	28	Devlin, et al., 1982
			LC-50	96	26	Devlin, et al., 1982
			LC-50	96	31	Devlin, et al., 1982
			LC-50	96	30	Devlin, et al., 1982
			LC-50	96	12.6	Pearson, et al., 1979
			LC-50	96	26.2	Geiger, et al., 1986
			LC-50	96	14.6	Geiger, et al., 1986
			LC-50	96	36.2	Geiger, et al., 1986
			LC-50	96	77.4	Mayes, et al, 1983
			LC-50	96	56.4	Mayes, et al, 1983
			LC-50	96	54	Mayes, et al, 1983

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
INVERTEBRATES						
Copper	7440-50-8	*Lymnacea luteola*	LC-50	96	0.172	Mathur, et al., 1981
		Acroneuria lycorias	LC-50	96	8.3	Warnick and Bell, 1969
		Pteronarcys sp.	LC-50	NR	50	Goettl, et al., 1972
		Ephemerella subvaria	LC-50	96	0.32	Warnick and Bell, 1969
		Chironomus tetans	LC-50	96	0.017	Gauss, et al., 1985
		Daphnids	LC-50	46	17	Mount & Norberg, 1984
		Daphnia magna	LC-50	96	0.01	Cairns, et al., 1978
		Daphnia pulex	LC-50	96	0.01	Cairns, et al., 1978
		Daphnia magna	LC-50	48	0.054	Mount & Norberg, 1984
		Daphnia pulex	LC-50	48	0.053	Mount & Norberg, 1984
		Ceriodaphnia reticulata	LC-50	48	0.017	Mount & Norberg, 1984
		Simocephalus vetulus	LC-50	48	0.052	Mount & Norberg, 1984
Copper (II)[c]		*Asellus aquaticus*	LC-50	96	9.21	Martin and Holdich, 1986
		Crangonyx pseudogracilis	LC-50	96	1.29	Martin and Holdich, 1986
Silver	7440-22-4	*Aplexa hypnorum*	LC-50	96	0.083	Holcombe, et al., 1987
		Tanytarsus dissimilis	LC-50	48	0.42	Holcombe, et al., 1987
		Daphnids	LC-50	48	0.011	Mount & Norberg, 1984
		Daphnia pulex	LC-50	48	0.014	Mount & Norberg, 1984
		Ceriodaphnia reitculata	LC-50	48	0.011	Mount & Norberg, 1984
		Simocephalus vetulus	LC-50	48	0.015	Mount & Norberg, 1984
Silver (I)[c]		*Crangonyx pseudogracilis*	LC-50	96	0.005	Martin and Holdich, 1986
Zinc	7440-66-6	*Dapnids*	LC-50	48	0.068	Mount & Norberg, 1984
		Daphnia magna	EC-50	48	1.1	Berglind and Dave, 1984
			LC-50	96	0.12	LeBlanc, 1984
			LC-50	48	0.068	Mount & Norberg, 1984
		Daphnia pulex	LC-50	48	107	Mount & Norberg, 1984
		Ceriodaphnia reticulata	LC-50	48	0.076	Mount & Norberg, 1984
		Dugesia tigrina	LC-50	96	7.4	See, et al., 1974
Zinc(II)[c]		*Lymnaea luteola*	LC-50	96	4.7	LeBlanc, 1980a
		Asellus aquaticus	LC-50	96	18.2	Martin and Holdich, 1986
		Crangonyx pseudogracilis	LC-50	96	19.8	Martin and Holdich, 1986
Chlorobenzene	108-90-7	*Daphnia magna*	LC-50	24	310	Bringmann and Kuhn, 1977
			LC-50	24	140	LeBlanc, 1980b
			LC-50	48	86	LeBlanc, 1980b

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
			LC-50	48	10.7	Cowgill, et al., 1985
			LC-50	48	13	Cowgill, et al., 1985
			LC-50	48	11.5	Cowgill, et al., 1985
			LC-50	48	8.6	Cowgill, et al., 1985
			LC-50	48	12.8	Cowgill, et al., 1985
			LC-50	48	15.4	Cowgill, et al., 1985
			LC-50	48	12.9	Cowgill, et al., 1985
			LC-50	48	21.3	Cowgill, et al., 1985
			LC-50	48	10	LeBlanc, 1980b
		Ceriodaphnia dubia	LC-50	48	11.8	Cowgill, et al., 1985
			LC-50	48	11	Cowgill, et al., 1985
			LC-50	48	10.4	Cowgill, et al., 1985
			LC-50	48	11.1	Cowgill, et al., 1985
			LC-50	48	7.9	Cowgill, et al., 1985
			LC-50	48	11.4	Cowgill, et al., 1985
			LC-50	48	8.9	Cowgill, et al., 1985
Napthalene	91-20-3	*Daphnia magna*	LC-50	24	6.6	Crider, et al., 1982
			LC-50	24	13.2	Crider, et al., 1982
			LC-50	24	17	LeBlanc, 1980b
			LC-50	48	8.6	LeBlanc, 1980b
			LC-50	48	3.4	Crider, et al., 1982
			LC-50	48	4.1	Crider, et al., 1982
			LC-50	48	2.16	Millemann, et al, 1984
		Daphnia pulex	LC-50	48	2.92	Geiger and Buikema, 1982
			LC-50	96	1	Trucco, et al., 1983
		Chironomus tetans	LC-50	48	2.81	Millemann, et al., 1984
		Physa gyrina	LC-50	48	5.02	Millemann, et al., 1984
		Nereis arenaceodentata	LC-50	96	3.8	Rossi and Neff, 1978
		Somatochlora cingulata	LC-50	96	1	Correa and Coler, 1983
PCB-1254	11096-82-5	*Daphnia magna*	LC-50	336	0.0018	Nebeker and Puglisi, 1974
			LC-50	504	0.031	Nebeker and Puglisi, 1974
			LC-50	504	0.0013	Nebeker and Puglisi, 1974
			LC-50	336	1.8	USFWS, 1986
			LC-50	504	1.3	USFWS, 1986
		Tanytarsus dissimilis	LC-50	504	0.00045	Nebeker and Puglisi, 1974

TABLE A-2
CIBA Cranston Facility
Aquatic Toxicity Data for Indicator Constituents of Concern - Pawtuxet River

Analyte	CAS No.	Test Species	Test Endpoint	Test Duration (hrs)	Endpoint Value (mg/L)	Reference
			LC-50	504	0.00065	Nebeker and Puglisi, 1974
		Ischnura verticalis	LC-50	96	0.2	Mayer, et al., 1977
			LC-50	96	0.2	Johnson and Finley, 1980
			LC-50	96	0.2	Stalling and Mayer, 1972
			LC-50	96	0.2	USFWS, 1980
		Ischnura spp.	LC-50	96	0.2	Johnson and Finley, 1980
		macromia sp.	LC-50	168	1	Mayer, et al., 1977
			LC-50	168	1	Stalling and Mayer, 1972
			LC-50	168	0.8	Johnson and Finley, 1980
			LC-50	168	0.8	Johnson and Finley, 1980
			LC-50	168	0.8	USFWS, 1980
Tinuvin 328		*Daphnia sp.*	EC-50	24	100	Ciba Geigy, 1994
Toluene	108-82-1	*Daphnia magna*	LC-50	24	310	LeBlanc, 1980b
			LC-50	24	470	Bringmann and Kuhn, 1977
			LC-50	48	310	LeBlanc, 1980b
			LC-50	48	313	USEPA, 1978

NOTES

NR: Not Reported

EC-50: Medial effective concentration

LC-50: Median lethal concentration

[c] Oxidation state in parentheses

References for Appendix A
Aquatic Toxicity Data - Pawtuxet River

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